

Attachments to Dr ST Mead Report

25 February 2021

Attachment 1

The Sustainability of Marine Farming in Beatrix Bay, Marlborough Sounds

March 2001

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Cover picture: The view of a Greenshell™ mussel farm from above and below (inset). (Extracted from www.greenshell.com.)

The Sustainability of Marine Farming in Beatrix Bay, Marlborough Sounds.

March 2001

Prepared For

**The Marlborough Sounds Trust,
New Zealand**

Prepared by

Shaw Mead¹, Kerry Black¹, & Andrew Longmore²

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Marine and Freshwater Consultants

P. O. Box 13048, Hamilton, New Zealand.
Ph. +64 7 858 5036, Fax.+64 7 858 5030
e-mail asrltd@clear.net.nz Internet www.asrltd.co.nz

2



PO Box 114, Queenscliff, Victoria
Australia.
Ph. +61 3 5258 0334
e-mail Andy.Longmore@nre.vic.gov.au

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Executive Summary

After reviewing the scientific literature and the evidence presented in relation to the coastal permit to Kuku Mara Partnership for a 42.25 ha mussel farm in west Beatrix Bay, it is our opinion that:

- current speeds and flushing rates are low in the Bay, in relation to carrying-capacity
- current measurements are inadequate for determining the long- and medium-term dynamics of the water body at the mussel farming sites.
- in the absence of full calibration, the numerical modelling output used in evidence by James (2000) is not useful for making sound judgments about carrying capacity
- modeling of complex eco-systems is in its infancy, lacking peer reviewed publication and solid calibration. The modelling results are therefore potentially misleading.
- food supply in West Beatrix Bay may not be adequate for existing farms, much less for additional undertakings.
- while the consistent occurrence of “adequate food” is probably the single most important factor determining the suitability of the Marlborough area for mussel culture, food sources vary between embayments and within mussel farms.
- the concurrence of slow current speed and high stock density will accentuate the depletion of food resources within the mussel farm and downstream.
- the proposed large farm may impact on other farms, and therefore on Bay eco-system health.
- potential effects of the long-term nutrient cycle on benthic communities may occur and that this matter has not been adequately addressed in evidence.
- enhanced ammonium excretion from a large-scale mussel farm could lead to a shift in the phytoplankton community to less edible species and that this matter has not been adequately addressed.
- flow reduction in the large farm could cause significant depletion of chlorophyll, and that fundamental information is lacking, suggesting a precautionary approach.

- provision to reduce the number of longlines on the new farm is absent if the monitoring shows that stage 1 is having a negative impact on chlorophyll levels in the bay.
- mussels are already stressed in Beatrix Bay, and therefore so is the natural biota in this environment.
- spatfall and harvest data for Beatrix Bay show significant fluctuations and downward trends that are indicative of stressed mussels at the present stocking levels. These trends are not consistent with the chlorophyll levels, which are being used as the main indicator for carrying capacity and monitoring of impacts.
- the estimates of carrying capacity given by James are inadequate as a basis for increasing stocking levels in Beatrix Bay.
- the West Beatrix Bay mussel farm is likely to negatively impact on the bay's ecology and on the existing mussel farms, contrary to the purpose and principles of the Resource Management Act (1991).

1 Introduction

1.1 Background

At a meeting held on 23 May 2000, the Hearings Committee of the Marlborough District Council granted a coastal permit to Kuku Mara Partnership for a 42.25 ha mussel farm in west Beatrix Bay – a resource consent for a similar sized farm in east Beatrix Bay was not granted (Fig. 1). However, a group of marine farmers in the Marlborough Sounds area, formally known as the Marlborough Sounds Trust, considered the evidence presented in the consent application to be inadequate as the basis for the decision. In addition, through over two decades of experience in Greenshell™ Mussel farming in the area, they believe that there are already indications of environmental stress in Beatrix Bay and that the addition of a large open-water mussel farm would exacerbate the problem. The review presented here considers the existing information supplied by Kuku Mara for the Resource Consent Application, scientific literature on the Marlborough Sounds marine environment and relevant international studies, in order to develop an opinion on the likely impacts of the proposed 42.25 ha mussel farm on the water quality of Beatrix Bay, and hence the wider marine environment.

1.2 Scope of Works

The Marlborough Sounds Trust commissioned ASR Limited to:

- undertake an international and local literature review of the impacts of mussel farming on water quality;
- analyse the present situation at Beatrix Bay, and;
- critically review the evidence of M. James (2000).

In order to:

- make an independent scientific assessment to develop an opinion on whether further challenge of the west Beatrix Bay coastal permit is justified and what further information would be needed to develop a case.



Figure 1a. Location map of Beatrix Bay in the Marlborough Sounds on the upper South Island.

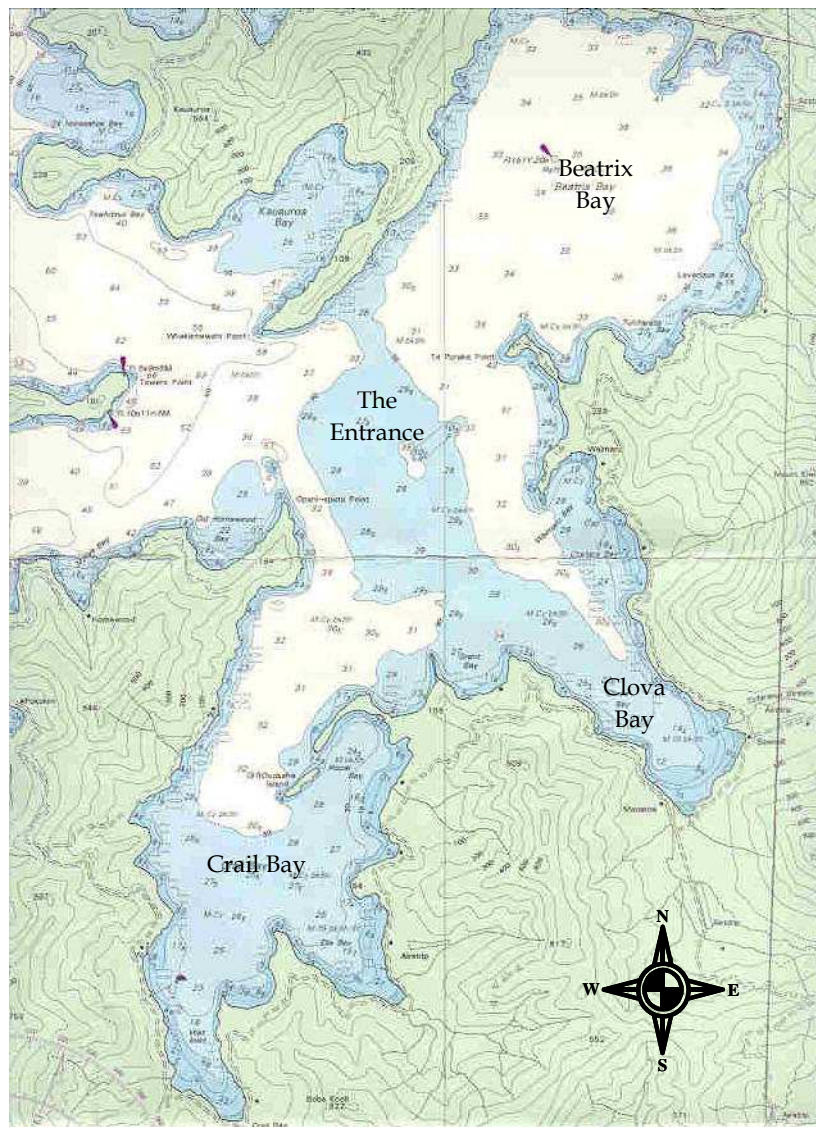


Figure 1b. The three regions of the Beatrix Bay side arm of Pelorous Sound the entrance region, Beatrix Bay to the north and Crail/Clova Bays to the south.

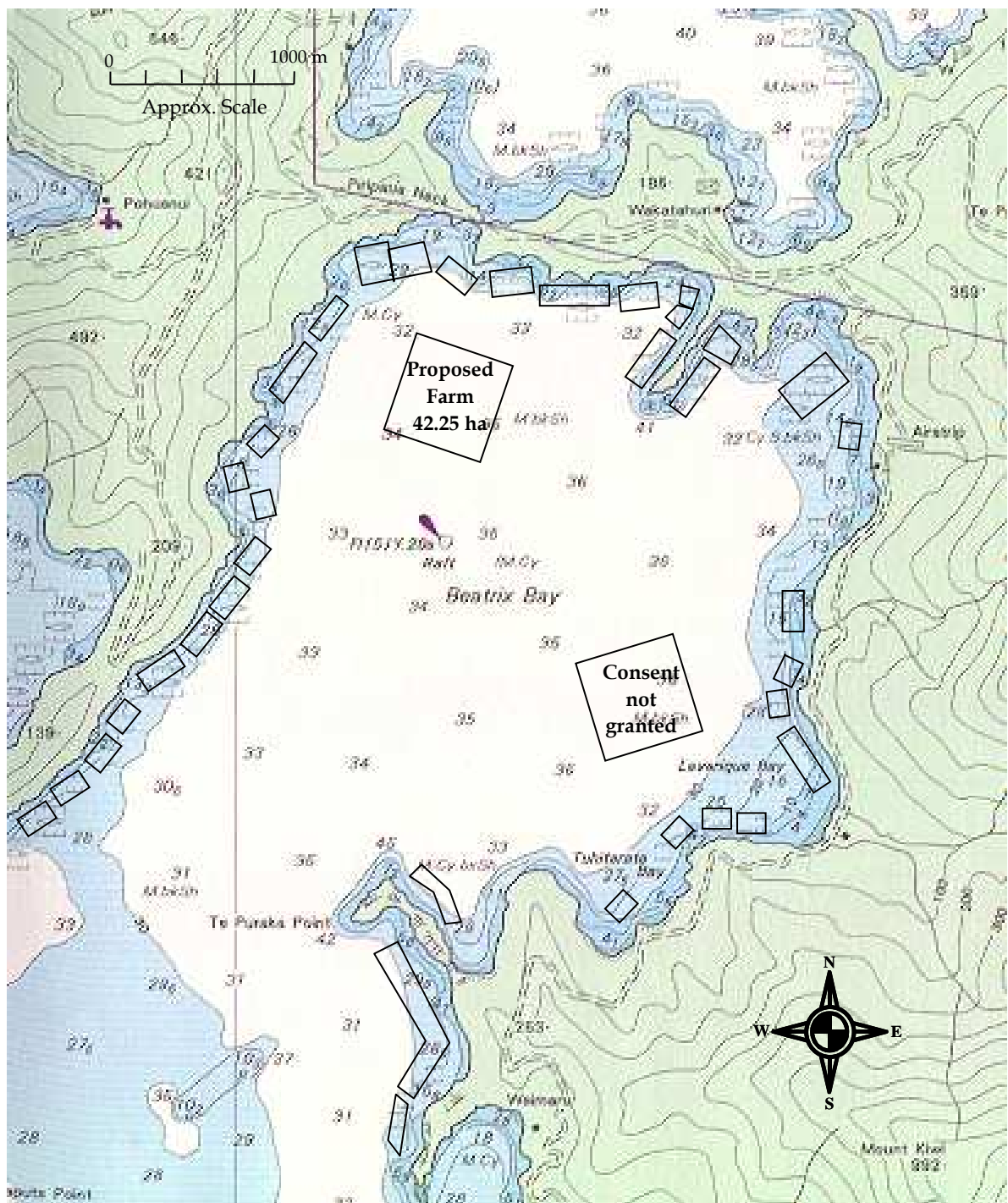


Figure 1c. Location map of the proposed marine farm in west Beatrix Bay, the existing marine farm positions and the position of the unsuccessful application for the east Beatrix Bay farm.

1.3 Sources of Information

International literature was obtained from a number of sources including Scientific Journals, consultant reports and data records, the World Wide Web, CDROM databases and unpublished theses. In addition, the AEE reports (Kuku Mara Partnership, 2000; Cawthron, 1999) and evidence presented to the environment hearing (e.g. James, 2000) were considered and compared to the literature review findings to evaluate the degree of certainty of the previous assertions.

1.4 Report Focus

While there may be other contentious issues arising from the Marlborough District Council's granting of a coastal permit for a 42.25 ha marine farm in western Beatrix Bay (e.g. recreational usage, aesthetic impact, etc.), the issues of ecological values, including marine habitats and sustainability, are of greatest concern to the impacts on the existing marine farms and the marine environment. Ecological impacts due to the addition of large offshore marine farms in Beatrix Bay were mostly addressed by James at the environment court hearing (James, 2000). In reaching its decision that the impact of the proposed new farm is "likely to be no more than minor", the resource consent hearing accepted James' assertions that:

1. Carrying capacity is greater than the sum of current and proposed new production;
2. Benthic impact of a new farm is likely to be small;
3. Staging will avoid an impact on other farms, and;
4. Mussels are good indicators of overall bay ecological health.

These are therefore the points that we will concentrate on. In addition, the effects of reduced water flows due to marine farm structures, with special attention to the physical setting of west Beatrix Bay, are addressed in relation to the understanding of water and nutrient flows.

2 Circulation and flushing

Linked to Cook Strait, Pelorus Sound is a drowned river valley system about 55 km long, with several side arms and bays. Beatrix Bay is one of three water bodies forming one side arm (Figure 1a). The three regions are the entrance region, Beatrix Bay and Crail/Clova Bays (Figure 1b). Beatrix Bay is roughly circular (about 4.5 km diameter) and mostly 30-35 m deep (Figure 1c).

The dominant circulation in the Sound (which affects Beatrix Bay) is a saline, nutrient-rich inflow at the seabed from Cook Strait and an outflow of lower salinity water at the surface. The Pelorus River is identified by James as being largely responsible for the strong and persistent stratification found in Beatrix Bay.

Spatial differentiation of the Marlborough Sounds into areas promoting high, medium and low condition indicates that some embayments are more suitable for mariculture than others (Waite, 1989). A fundamental aspect of good mussel farm location is high current flow. High currents provide a larger volume of water to filter food from, reduce impacts on the seabed and mix the locally-high chlorophyll levels. Regions of slow currents are more likely to incur benthic impacts and receive less food.

Water circulation in Beatrix Bay is said to follow a clockwise pattern, with water from West Beatrix probably reaching Laverique Bay some 2 weeks later (Sutton and Hadfield, 1998, unpublished data – cited Ross *et al.*, 1998). However, evidence for such a simple pattern within the Bay is not provided by the current meter measurements made by NIWA (Kuku Mara Partnership, 2000).

From James' evidence and other sources, the following are evident:

- Instantaneous currents at the East and West Beatrix Bay sites are only a maximum of about 10-12 cm.s⁻¹, presumably at peak tidal flow.
- Currents in the water column at the mussel farming sites are only 30-100 m/hour, i.e. 8 mm.s⁻¹ to 2.7 cm.s⁻¹ when averaged over time.
- Time-averaged flow directions through most of the water column are highly variable and not strongly biased towards a predominant direction.
- Re-circulating eddies occur.
- All directions of time-averaged flow have nearly equal probability of occurrence through much of the water column, except near the seabed.
- The small cumulative flow vectors suggest that a significant proportion of the water leaving the site would be expected to return again, resulting in slow flushing rates.

- Drogues released from the mussel farming sites were caught in internal eddies and one ended up basically where it started.

It is our opinion that the current speeds and flushing rates are low in the Bay, in relation to carrying-capacity (see further discussion below).

The duration of the current measurements made by NIWA is too short to be certain about long-term trends or seasonal patterns. James cites bottom-mounted ADP current profiler measurements at the east Beatrix Bay site over autumn 1998 (18 February – 24 April) and for 10 days only at the west Beatrix Bay site (September, 1999). Only limited drogue tracking at 2-8 m depth was undertaken. The duration of the ADP boat-mounted measurements is not given.

It is our opinion that the current measurements are inadequate for determining the long- and medium-term dynamics of the water body at the mussel farming sites.

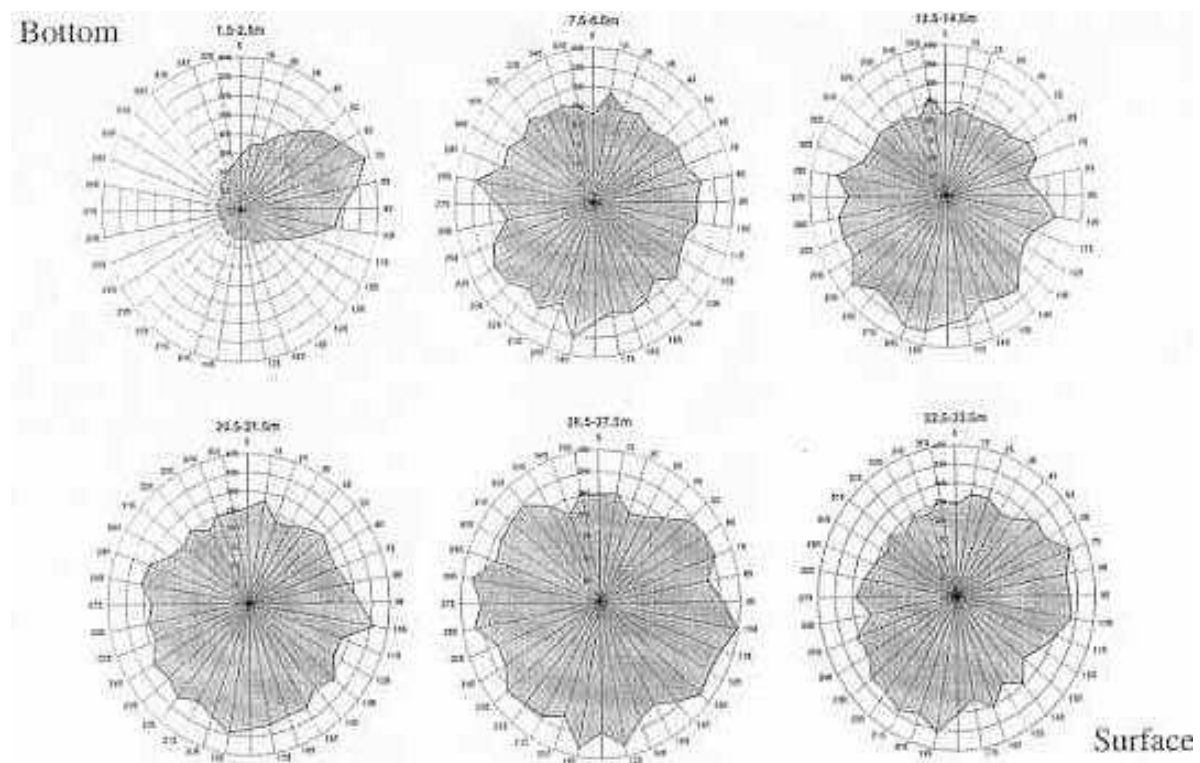


Figure 2. (MRJ8) The direction of currents at various heights above the seabed at the East Beatrix Bay site, from evidence of James. He notes that a wide range of current directions occurs, except close to the seabed.

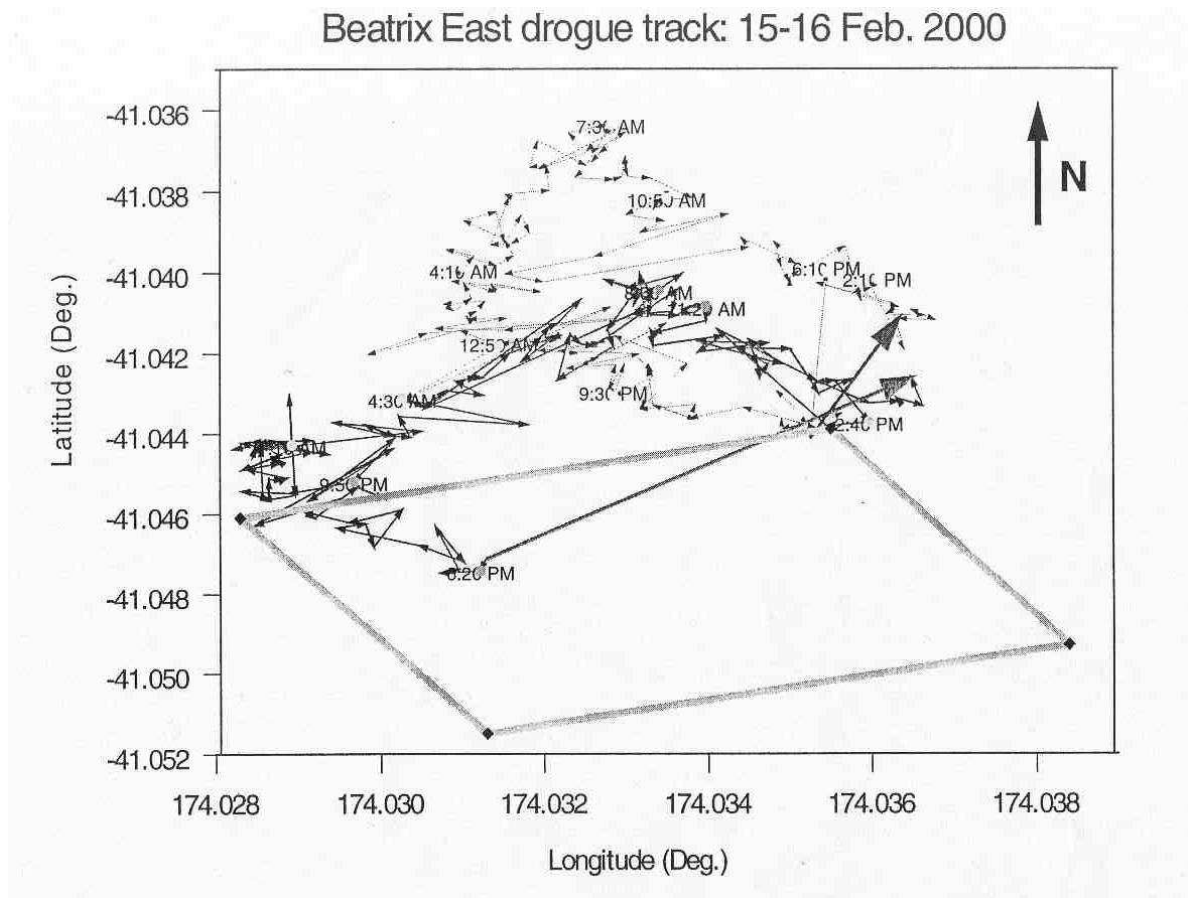


Figure 3. (MRJ12) Drogue tracks at the East Beatrix Bay site, from the evidence of James. He noted that both drogues were affected by an eddy.

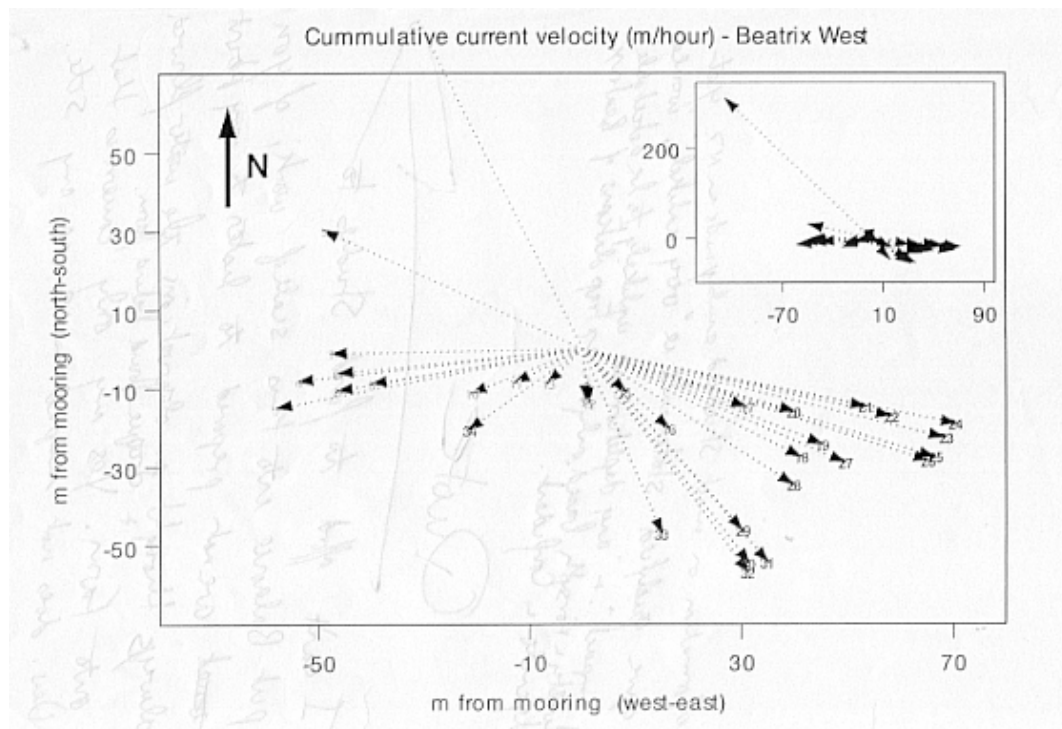


Figure 4. Current flow vectors at West Beatrix Bay, time averaged over the record duration (m/hour) (from the evidence of James).

3 Carrying Capacity

Eco-system numerical models are, by their nature, gross simplifications of naturally complex systems. We can have no confidence in the output (predictions) of models unless and until they can be shown, as a minimum, to reproduce field observations. NIWA has chosen to model only selected broad functional categories of the ecosystem (Ross *et al.* 1999). The NIWA model, from which carrying capacity for Beatrix Bay was estimated, includes modules to reproduce hydrodynamics, phytoplankton growth and nutrient dynamics, and mussel growth and condition (NIWA 2000a). Ross *et al.* (1999) stated, “The applicability of such models for management decisions, when not calibrated and tested, is questionable”. No evidence has yet been produced that the NIWA model accurately reproduces any of these variables. At the time of James’ submission that current stocking in Beatrix Bay is well below the carrying capacity, NIWA (2000a) admitted that they were still validating the model. Certainly no details of model calibration have been published for peer review. We believe that some model parameters used in the NIWA model are inherently inaccurate, and other important parameters have been left out completely. The hydrodynamic model is described as consisting of “6 separate boxes”. As admitted by Hadfield (pers. comm.), the box model had limitations.

It is our opinion that in the absence of full calibration, the model output is not useful as evidence or for making sound judgments about carrying capacity

Dowd (1997) warned that simple ecological models of cultured bivalve growth are limited in their predictive capacity because of high sensitivity to small changes in physiological parameters describing the mussel energy budget. Grant and Bacher (1998) came to a similar conclusion, using the example of the digestibility of food. In Canada, the ratio of particulate organic carbon to particulate organic matter (POC/POM) varies naturally by 300% during the mussel growth cycle. However, in the models they developed, a 10% change in this ratio (digestibility) led to a 40-60% change in predicted mussel weight after 8 months. The situation may be even more uncertain with the NIWA model, since POC/POM varied by more than 500% over 24 hours in Kenepuru Sound (Hawkins *et al.* 1999).

Grant and Bacher (1998) found that total chlorophyll measurements do not account for temporal changes in the composition of phytoplankton communities and their differing digestibility. They concluded (and Campbell and Newell 1998 concurred) that food quality is more important than quantity in accurately predicting growth. Waite (1989) found a similar trend with the quality of food in the diet of *P. canaliculus* (Greenshell™ Mussels), as did Prins *et al.* (1994) with *Mytilus edulis* (Blue mussel). Hawkins *et al.* (1999) found that retention efficiency of organic matter and chlorophyll each varied strongly with both the abundance and composition of available seston, and accurate estimation of both were critical

to the development of further model parameters. As far as we can tell, the NIWA model uses total chlorophyll with a fixed digestibility, and takes no account of varying availability or organic content. Ogilvie *et al.* (2000) stated that the relative nutritional value of phytoplankton from different depths in Beatrix Bay is unknown, but NIWA assumes the plankton in deeper water to be accessed by the proposed farm has the same nutritional properties as that in the surface layer.

Campbell and Newell (1998) believed that “an accurate determination of the carrying capacity for mussel aquaculture within the larger ecosystem would require the modelling of system components such as seaweed, zooplankton, macrobenthos, wild mussels, etc.”. Zooplankton appears to be the only one of these components included in the NIWA model. Grant *et al.* (1998) found that mussel fouling comprised about 25% of the nutritional demand of a mussel farm, but there is no evidence the NIWA model takes such a large demand into account. Indeed, the experiments of mussel growth on which the NIWA model is based (Hawkins *et al.*, 1999) were carried out with mussels that had been cleaned of all epibiotic growth. Odum *et al.* (1983), in one of the earliest simulations of mussel culture in Marlborough Sounds, predicted that stocking at a level to maximise profits would lead to a reduction in wild mussels, which are the source of the spat necessary for mussel culture. There is already evidence of poor and variable spat fall at current stocking levels. Spatfall statistics in Beatrix Bay (see below) and the studies of Bayne (1976) show spatfall reductions, possibly due to stressed mussels during prolonged time periods resulting in an increase in abnormal embryonic development.

Variables modelled by Hawkins *et al.* (1998) as inputs to the NIWA mussel nutrition sub-model included total organic content, clearance rate, retention efficiency and net absorption rate. Between 34 and 47% of the variance in these variables was not explained by the models derived. In other words, a large proportion of the natural variability of measures of mussel food supply and growth is not accounted for in model derivation. In addition, all of these relationships were derived for mussels of a single size; we cannot assume that the same relationships derived for one size apply over the whole growth cycle. For example, Waite (1989) found that the maximum growth efficiency of mussels in the 30 mm length class was 0.48 compared to 0.66 in the 80 mm length class.

It is our opinion that the modelling is in its infancy, lacking peer reviewed publication and solid calibration. The modelling results are therefore potentially misleading.

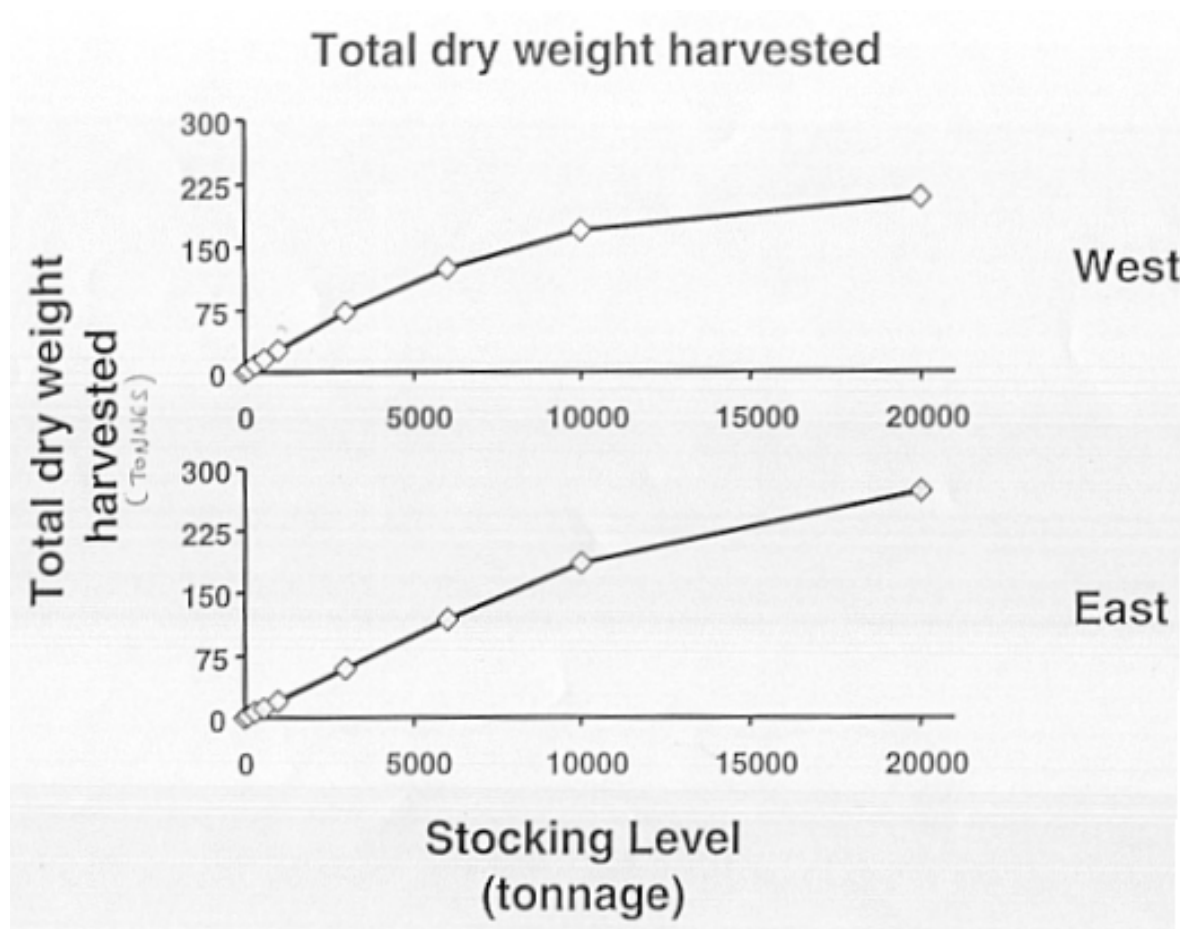


Figure 5. Model estimations of harvest weight vs stocking level, submitted as evidence by James (MRJ20).

James' exhibit MRJ 20 (estimated harvest weight vs stocking level) indicates that stocking in east Beatrix Bay could be increased to at least 10,000 tonnes before yield declined (Fig. 5). In contrast, some curvature is predicted for west Beatrix Bay (Fig. 5), even at current stocking levels. This implies that stocking in the east Beatrix Bay could increase significantly without affecting current production, whereas additional production in the western bay will affect current producers (by extending the time taken to grow). Nevertheless, the tribunal rejected the application for the eastern farm, because they had "sufficient information to say there would only be sufficient nutrients and food to sustain one farm". MRJ20 indicates farms on the western side are more vulnerable to impact from additional production than those on the eastern side.

Hawkins *et al.* (1999) observed wasting of mussels at chlorophyll concentrations below $0.86 \mu\text{g L}^{-1}$, and no significant growth below $1 \mu\text{g L}^{-1}$. Similarly, the critical chlorophyll concentration has been found to be between $1 \mu\text{g L}^{-1}$ (Ross *et al.*, 1998) and $1.5 \mu\text{g L}^{-1}$

(Waite, 1989) in the Pelorous Sound, below which mussel condition is likely to decline. Given that there have already been substantial periods during 1996-98 of chlorophyll concentration below $1 \mu\text{g L}^{-1}$ in Beatrix Bay (Ross *et al.*, 1998), perhaps driven by meteorological or climate change factors, we can have no confidence that the food supply will be adequate for existing farms, much less additional ones. The idea of therefore supporting increased stocking is difficult to justify. Low mussel condition has frequently been the case at existing farms in the west Beatrix Bay area and growth rates and yields have regularly been depressed (see below).

It is our opinion that food supply in West Beatrix Bay may not be adequate for existing farms, much less for additional undertakings.

Regions of slow currents are more likely to incur benthic impacts and receive less food. Waite (1989) identified slow current speed, low food concentration, high salinity and high temperature as factors that may limit feeding and growth of *P. canaliculus*. Dense communities of *P. canaliculus* clear substantial proportions of the food flowing past farmed populations, and local depletion of food limited both its food intake and growth (Waite, 1989). Waite (1989) found that food depletion appeared to occur when rates of food consumption approached rates of influx of food, and this balance between food supply and consumption may limit the maximum viable size of farm communities. Farm communities become isolated from their food supply during periods of slow current flow, and adequate flow of water through an embayment is essential for successful mussel cultivation.

Farms may therefore become isolated from food-bearing currents and as a result they are likely to provide sub-optimal conditions for intensive mussel culture (Waite, 1989). Stronger current flows have been recorded in major channel systems within the Marlborough Sounds (Heath, 1982 – cited Waite, 1989) and these channels may represent areas that can support high stock densities and rapid growth. However, the embayments are subject to much slower currents and are therefore less able to sustain high-density farming. Indeed, depletion is more likely to be noticeable on farms sited in low current flows (NIWA, 2000b). There was a measured 18% decline in tissue content for mussels in the middle of Crail Bay farms (Waite, 1989), and larger farms would be expected to have a greater impact.

For example, current speed is a vital factor in dense cultures of *M. edulis* (Rosenburg and Loo, 1983 - cited Waite, 1989), and Waite's (1989) study proved that flow inhibition occurs in the mussel farms in Marlborough. Presence of culture ropes retarded flow of water and extended the period that water was grazed by mussels. At current speeds of $2\text{-}10 \text{ cm.s}^{-1}$, *P. canaliculus* consumed 15-60% of available food. These flows are very similar to the current speeds in West Beatrix Bay, which attain up to about 12 cm.s^{-1} , but the mean currents averaged through time are more like $1\text{-}2 \text{ cm.s}^{-1}$, in the absence of the farm. Currents would be significantly reduced once the farming is established. Measurements have shown that

flows are deflected and run parallel to longlines. Current vectors parallel to longlines were translated into flow, whereas transverse vectors were absorbed by longlines (Waite, 1989). Sullivan's (1978) model of fluid striking an impermeable plane could be used to describe the flow past longlines. The close proximity of longlines and small spacing between culture ropes in mussel farms in New Zealand may accentuate the magnitude of flow inhibition, and therefore pose a significant constraint on productivity (Waite, 1989).

Waite (1989) identified the following factors that may regulate the transport of food through mussel farms, and should be investigated further:

1. angles between longlines and prevalent currents,
2. separation between adjacent pairs of long lines,
3. length and depth of longlines, and,
4. stock density, size and distribution.

Adequate transport of externally produced food is needed to supply food to farmed mussels. Without adequate flow, farms can deplete food from substantial areas of the Sounds and reduce the concentration of phytoplankton available to other grazers. Waite (1989) advocated redesign of mussel farms to reduce both the attenuation of currents and the depletion of food should also enhance the growth of mussels and protect the habitats of other grazers. The impacts on food transport through a 42.25 ha marine farm structure in west Beatrix Bay have not been adequately addressed. It has been indicated that 'there is currently no information on how farms, even 3 ha farms, affect the current flow' (NIWA, 2000b), however, our literature search has provided several examples that show significant reductions in current speed.

While the consistent occurrence of "adequate food" is probably the single most important factor determining the suitability of the Marlborough area for mussel culture, food sources vary between embayments and within mussel farms. It is our opinion that the concurrence of slow current speed and high stock density will accentuate the depletion of food resources within the mussel farm. The currents in West Beatrix Bay are very small and so at least 15-60% of food could be consumed within the farm. West Beatrix Bay is subject to low current flows and therefore limited food supplies.

4 Benthic Impacts

James used a video showing sea stars feeding on fallen mussel clumps to base his assertion that the proposal would have a minimal impact on the benthos. Similarly, the Cawthron (1999) site assessment mainly targets abundances and types of species. However, the major impacts are likely to be somewhat less visible. They may include changes to nutrient recycling from the sediment, and changes to sediment fauna. Mussel culture leads to enhanced sedimentation of organic matter (Barranguet 1997), and the rate of sedimentation is linearly related to chlorophyll biomass (Hatcher *et al.* 1994). In a general sense, increased organic enrichment leads to a reduction in the number of species living in the sediment, but possibly an increase in total abundance of a few opportunistic species, usually marine worms (Pearson and Rosenberg 1978).

Mitro *et al.* (2000) observed a significant decline in meiofaunal abundance under a mussel farm, though the implications of this to the wider ecosystem are not clear. Kaspar *et al.* (1985) found sediment under a small (1.5 ha) farm was enriched in organic nitrogen and ammonium, while the benthic community was depauperate compared to a control site. Denitrification (conversion of nitrate to N₂ gas) was ~20% higher in sediment at the mussel farm than at the reference site, and was 10 times higher in the detritus-covered mussels at the farm than in the reference site sediment. Enhanced denitrification leads to a net loss of nitrogen from the system. The extent of denitrification is a balance between the rate of supply of organic matter, its bacterial breakdown, and the supply of oxygen, which may be mediated by infauna irrigating the sediment (Berelson *et al.* 1998). The impact of increasing farm size (from 2-3 ha to 42 ha) on denitrification is unknown. If an increasing area of sediment enriched in organic matter below a much larger 42 ha farm has no impact on the infauna, the increased size may lead to increased denitrification (and loss of nitrogen from the ecosystem), which may impact on all farms in the Bay. Alternatively, the organic deposition over a wider area may cause a reduction or loss of the infauna, leading to a switch from denitrification to ammonium release. Whether this is a benefit to mussel production or not will depend on whether the ammonium is taken up by beneficial plankton, or those of low nutritional value. Bear in mind that sediment nutrients are released to the bottom waters, which are already nitrate-rich, and apparently inaccessible to surface-dwelling plankton in summer. However, if the sediment-released nutrients do stimulate a bloom of non-beneficial algae, the dominant bottom current will carry the nutrients (or bloom cells) toward the inshore farms. The large farm may well produce an impact on other farms, while escaping such impacts itself.

It is our opinion that the large farm may impact on other farms, and therefore on Bay ecosystem health.

In a Canadian study, the largest response of the sediment community to suspended mussel culture was increased ammonium release from the sediment year-round, with the highest rate in summer (Hatcher *et al.* 1994). There was a negative relationship between bottom water nitrate concentration and ammonium flux, which Hatcher *et al.* (1994) took to indicate ammonification (conversion of nitrate to ammonium) and denitrification (conversion of nitrate to N₂ gas), both processes reducing the concentration of the nitrogen species thought to be most critical for phytoplankton in Beatrix Bay.

Changes in sediment nutrient cycling in Beatrix Bay are important, because the nitrogen supplied by sediment release is of a similar magnitude to Cook Strait input, and much larger than river flow or mussel excretion inputs (Gibbs *et al.* 1992). Ross *et al.* (1999) confirmed that sediment nutrient recycling has a strong influence on the dynamics of the Beatrix Bay ecosystem.

NIWA (2000a) admits that, “increasing the yield also needs to be viewed in the context of other potential effects of the long-term nutrient cycle and benthic communities”. No such context was provided in James’ submission to the application process; they were neglected completely.

It is our opinion that potential effects of the long-term nutrient cycle on benthic communities may occur and that this matter has not been adequately addressed in evidence.

5 Water Column Impacts

Bradford *et al.* (1987) indicated that poor mussel condition was a periodic problem in the early 1980s, which was attributed to food shortage due to nutrient depletion in summer. Currently about 160 ha of Beatrix Bay is farmed, with 13 ha used for spat collection; the proposed farm increases the area by 24%. Ross *et al.* (1998) believe that inner Sound farms are more affected by natural hydrographic variation in nutrient supply than those closer to the Sound mouth. Indeed, the high concentrations of ammonium excreted by mussels during summer periods (Ogilvie *et al.*, 1998) is indicative of loss of mussel condition through protein catabolism due to food shortage (Bayne, 1976); i.e. the shellfish are stressed.

Ogilvie *et al.* (1998) observed occasions when chlorophyll concentration was higher inside farms than outside. They attributed this to phytoplankton growth enhanced by ammonium excreted by mussels. At first glance, this may be seen as an advantage, leading to higher food production, and therefore potentially higher mussel growth. However, Prins *et al.* (1994) noted that mussels reduce clearance rates if the available phytoplankton are not suitable food. Beatrix Bay is an ecosystem in which plankton were originally dependent on riverine and oceanic nitrate, and it is not surprising that growth of the endemic plankton, dominated by diatoms and dinoflagellates, is stimulated more by nitrate than ammonium (Gibbs and Vant 1997). However, an ecosystem in which increasing amounts of ammonium (from mussel excretion) become available may suit non-endemic phytoplankton species, which are unsuitable as a food source. For example, *Phaeocystis sp.*, the dinoflagellate *Gyrodinium aureolum* and the chrysophycean *Aureococcus anophagefferens* have all inhibited mussel filtration (Prins *et al.* 1994), possibly by clogging the gills with mucus. Rhodes *et al.*, (1995) reported *Phaeocystis* blooms in NZ waters in 1981. Coccolithophore blooms in 1992 were associated with fish mortalities in Big Glory Bay, NZ, and growth in culture was enhanced by ammonium addition. Similarly, a raphidophyte (*Heterosigma carterae*) bloom in Big Glory Bay in 1989 was associated with increased nitrogen supply (from fish farming), and has led to fish kills internationally (Chang and Page 1995).

The concern is therefore one of enhanced ammonium excretion from a large-scale mussel farm leading to a shift in the phytoplankton community to less edible species. This has not been addressed.

It is our opinion that enhanced ammonium excretion from a large-scale mussel farm could lead to a shift in the phytoplankton community to less edible species and that this matter has not been adequately addressed.

6 Staging

Staging can be considered with respect to:

- chlorophyll depletion

In his submission to the tribunal, James admitted that he cannot estimate depletion of food supply by such a large farm. All previous measurements have been on much smaller farms. One problem he faces is that water flow is restricted through even small farms (Waite, 1989). Gibbs *et al.* (1991), Boyd and Heaman (1998) and Karayucel and Karayucel (1998) observed reductions in flow through farms of up to 70% compared to currents outside the farms. Feeding efficiency (and therefore carrying capacity) will be lower under such conditions than if flow reductions are ignored. In addition, longlines have been found to be relatively impermeable to currents and effectively deflect currents to run parallel to them (Waite, 1989), the large scale farm proposed for west Beatrix Bay could have a large impact on the present circulation patterns in the western quarter of the bay. There does not appear to be any scope for the tribunal to reduce the stocking rate on the proposed farm, if the first stage is found to have a significant impact on the other farms. Ogilvie *et al.* (1998) found chlorophyll depletion of up to 72% in small farms. No one has been willing to estimate depletion in a farm 10-20 times larger, but James proposes simply to measure it once the farm is constructed. In the absence of such fundamental information, a precautionary approach may be warranted.

It is our opinion that flow reduction in the large farm could cause significant depletion of chlorophyll, and that fundamental information is lacking, suggesting a precautionary approach. There is no provision to reduce the number of longlines on the new farm if the monitoring shows that stage 1 is having a negative impact on chlorophyll levels in the bay.

- ammonium production

Mussels excrete ammonium, which may fuel increased plankton growth (Barranguet 1997). It is possible that ammonium produced by many small, widely-spaced farms is dispersed by mixing with water between the farms before relatively high concentrations confer a competitive advantage on one species over another. On the other hand, one large farm is more likely to produce a significant area of high ammonium concentration, making algal blooms more likely. No one is in a position to know whether such blooms would be of edible or inedible plankton.

In our opinion the potentially negative effects of significant areas of high ammonium concentration have not been addressed

7 Mussels as Indicators of Environmental Health

Mussel farms have a high biomass and biological activity, which indicate that cultivated *P. canaliculus* is a key element in the Marlborough ecosystem (Brinkhurst, 1974 – cited Waite, 1989). James asserted that “if mussels are doing well, other biota should also be doing well”. Hawkins *et al.* (1999) observed that wasting occurred for mussels supplied with less than 0.86 ug L⁻¹ chlorophyll, and that significant growth could only be expected for chlorophyll concentrations above 1 ug L⁻¹. Chlorophyll concentrations below 1 ug L⁻¹ were experienced in Beatrix Bay for considerable periods during 1996-98, during which periods mussels did not do well. By James’ definition, the Beatrix Bay ecosystem has been unhealthy for considerable periods. While the cause(s) of the poor growth may be natural oceanographic processes, there is no reason to believe the extra food consumption, which will arise from a 24% increase in farmed area will do anything but cause a further deterioration in mussel condition, and hence other biota in the area. Indeed, mussel farmers from the area believe that the marine environment in the area is already stressed. Since the beginning of NIWA’s research programme in the Pelorous Sounds there has been a rapid decline of the condition of the mussels grown in the area, and suggested that this is a possible indicator of a sustainable production problem (Ross *et al.*, 1999).

James predicted that mussels on the western side of Beatrix Bay will take ~10 weeks longer to grow to 100 mm than they do now if production increases to 6,000 t/a. Clearly mussels must be under stress if their growth rate slows; this is hardly a sign of good health.

It is our opinion that mussels are already stressed in Beatrix Bay, and therefore so is the natural biota in this environment.

8 Industry data

We have been provided with mussel spat fall and industry growth/tonnage data. Here we present this information graphically (Figs 6 & 7).

8.1 Spatfall Data

Spatfall data has been collected for the Pelorous Sound area since the mid-1980s (J. Jenkins, pers. comm.). However, detailed records of spat-catching sites in Beatrix and Clova Bays (both located in the same arm of the Sound – Figure 1) are only available back to the 1994-95 season. Figure 6 shows the total spatfall for Beatrix Bay (Fig. 6a) and Clova Bay (Fig. 6b).

Figure 6 shows the downward trend from 1994 to 2000 in spatfall numbers at the spat-catching sites in the Beatrix Bay arm of Pelorous Sound, which has recently been a cause of concern for people in the mussel industry. However, spatfall levels have recently increased (especially for the Clova Bay area – Fig 6b). Levels of spatfall were low for the present season up until the last week of April, when levels greatly increased (34% of the Beatrix Bay spatfall and 61% of the Clova Bay spatfall totals for this season occurred in the one week – J. Jenkins, pers. comm.). The fluctuations seen in the spatfall data are most likely due to natural oceanographic processes.

The depression in spatfall levels over the 1997-2000 period remains a concern, especially when viewed with the available chlorophyll data (see below). Several studies by Bayne (Gabbott & Bayne, 1973; Bayne 1975; Bayne, 1972; Bayne *et al.*, 1975 – all cited Bayne, 1976) have found that even though gonad development continues in stressed mussels, the mussel larvae from them develop abnormally and result in low spat yield. This has been linked to low energy reserves acquired from the poor condition adults (Bayne, 1976). While it may be argued that the reasons for the decrease in spatfall are due to natural oceanographic processes, these data indicate that there is a strong possibility that mussels are under environmental stress for prolonged periods even at the current stocking levels, particularly in adverse oceanographic conditions. Increasing stocking levels in Beatrix Bay will increase this stress on cultivated mussels and therefore the other naturally occurring biota in the area.

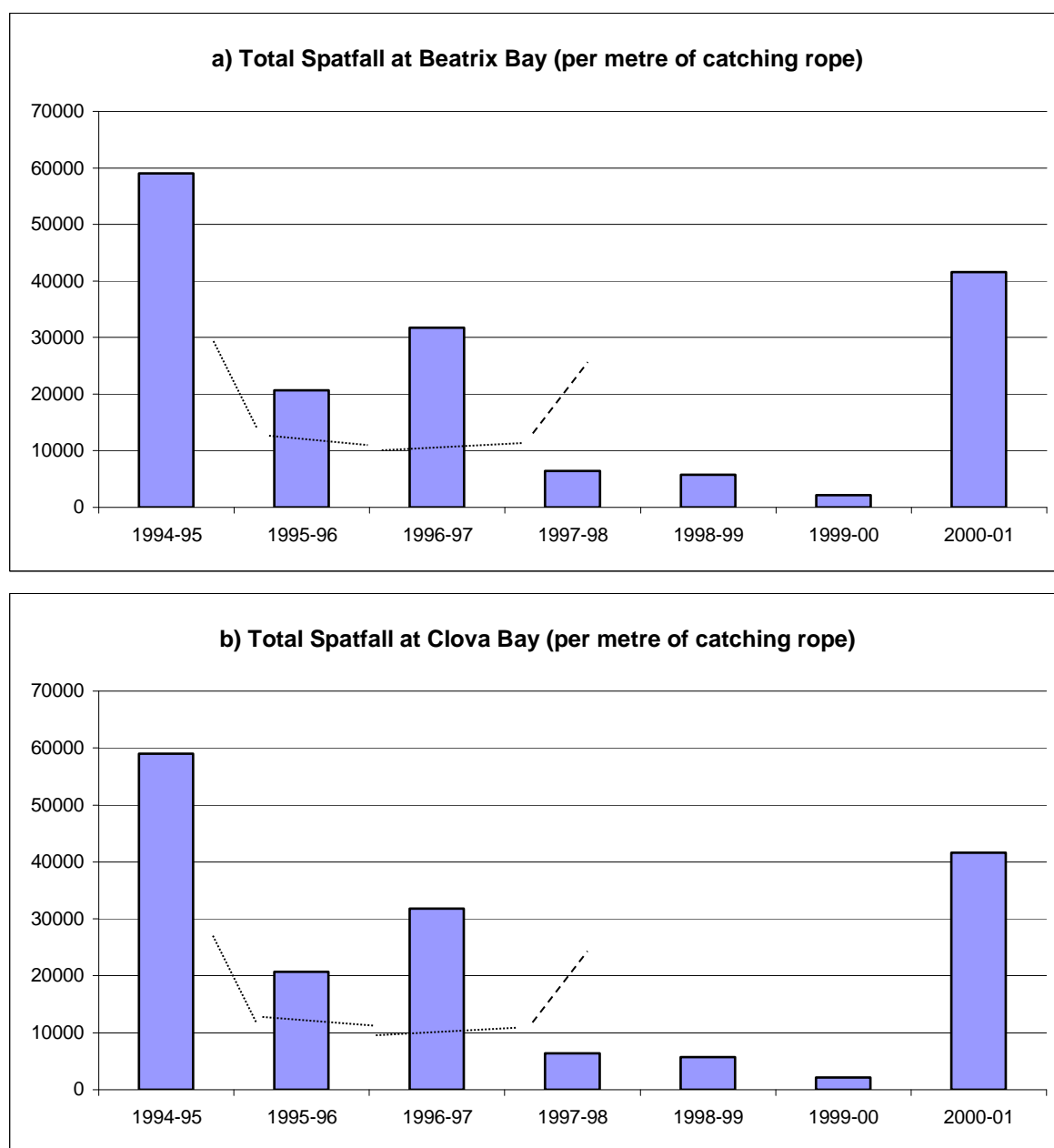


Figure 6. Total spatfall recorded at 10-15 m depth for Beatrix Bay (a) and Clova Bay (b) in Pelorous Sound. The total for 2000-01 is up to 30/4/01. Trend line of chlorophyll level represented by dashed line (not to scale – see Fig. 8)

8.2 Industry Growth Data

Harvest information provided for Sealords managed farms in Beatrix Bay is presented in Table 1. The most informative dataset in Table 1 is the average kilograms of mussels harvested per metre of longline seeded. While the nett harvest weight has fluctuated between 20,587 and 27,102 kg each year, the weight of mussels per metre of rope has followed a downward trend similar to that shown in the spatfall data (Fig. 6).

Table 1. Harvest information for Beatrix Bay (for Sealords managed farms only – A. Lumberg, pers. comm.).

Date	1994	1995	1996	1997	1998	1999	2000
Average of nett kg harvested	25130.9	24293.8	20587.2	26575.7	27102.6	21431.5	23399.7
Average of growth time	14.4	17.4	16.3	19.6	19.2	14.2	16.2
Average of metres seeded	3665.2	3474.2	3532.6	3731	4292.3	3492.3	3973.4
Average of kg/m seeded	7	7.6	6.4	8.3	6.2	6	5.3

When the phytoplankton chlorophyll-a level trends at Beatrix Bay (Ross *et al.*, 1998) shown in Figure 8 are compared to both the average kg/m seeded (Fig. 7) and the spatfall data (Fig. 6) for similar periods (March 1995-98) it is evident that the low levels of phytoplankton measured in 1996-97 do not correspond with low harvest per metre or low spatfall. On the contrary, the harvest per metre of rope and spatfall levels are elevated in 1996-97 (Figs. 6 & 7). This casts doubt on the predictions of stocking levels (in relation to food source) submitted by James (2000) and again points to concerns that the carrying capacity of Beatrix Bay may already be exceeded; spatfall and harvest per metre of rope decrease significantly in 1998 (Figs. 6 & 7), even though phytoplankton levels were found to be high (Figure 8). This may be particularly evident in years when the oceanographic conditions are not suitable for transporting the necessary volumes of food-rich waters into the bay.

It is our opinion that the spatfall and harvest data for Beatrix Bay show significant fluctuations and downward trends that are indicative of stressed mussels at the present stocking levels. These trends are not consistent with the chlorophyll levels, which are being used as the main indicator for carrying capacity and monitoring of impacts. The estimates of carrying capacity given by James are inadequate as a basis for increasing stocking levels in Beatrix Bay.

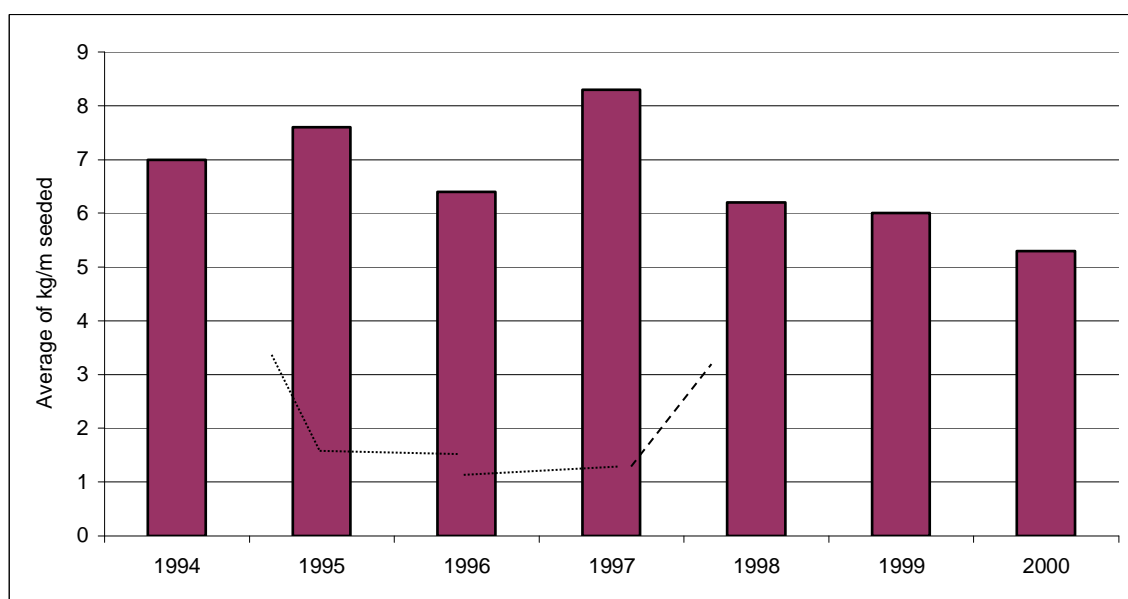


Figure 7. Average kilograms of mussels harvested per metre of longline seeded for Sealords managed farms in Beatrix Bay. Trend line of chlorophyll level represented by dashed line (not to scale – see Fig. 8)

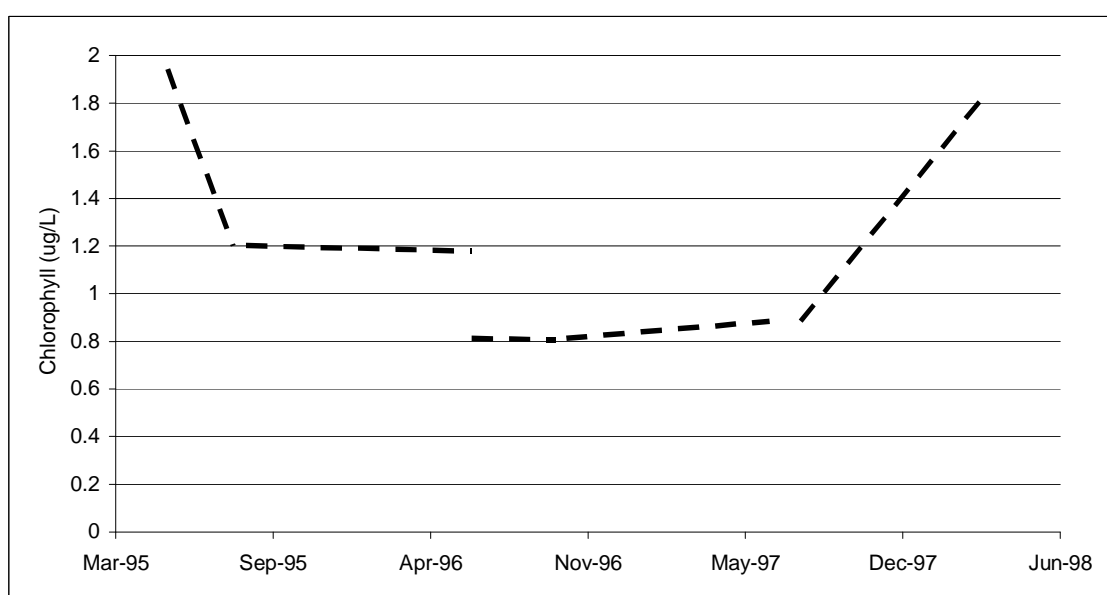


Figure 8. Trends in phytoplankton levels at Beatrix Bay (from Ross *et al.*, 1998).

9 Summary and Conclusions

Mussel farming is of great importance to New Zealand's export trade. In turn, the mussel industry is vitally dependent on a high quality marine environment, as fostered by the 'Purpose and Principles' of the Resource Management Act (1991) which are to promote the sustainable management of natural and physical resources (S5(1)) and "safeguarding the life-supporting capacity of air, water, soil and ecosystems" (Section 5(2)a, RMA (1991)).

The scientific literature and submissions by James both indicate that further stocking of mussels in Beatrix Bay will negatively impact on the health of the environment. Although of questionable validity, the numerical modelling has further demonstrated this contention. An additional 42.25 ha mussel farm in Beatrix Bay (a 24% increase in mussel farming within the embayment) is therefore not supportable. The western site appears to be less suitable than the eastern site.

It is our opinion that the West Beatrix Bay mussel farm is likely to negatively impact on the bay's ecology and on the existing mussel farms, contrary to the purpose and principles of the Resource Management Act (1991).

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Attachment 2

**Desktop Summary of Current Level of the Science and
Understanding of the Cumulative Ecological Impacts of
MusselFarms Ring-Fencing Coastlines such asBeatrix Bay,
Marlborough Sounds**

Desktop Summary of Current Level of the Science and Understanding of the Cumulative Ecological Impacts of Mussel Farms Ring-Fencing Coastlines such as Beatrix Bay, Marlborough Sounds



Prepared for:
The Pelorus Boating Club & The Kenepuru and
Central Sounds Residents Association



eCoast Ltd
Marine Consulting and Research
PO Box 151
Raglan
New Zealand

Telephone: +64 21 423 224
Email: info@ecoast.co.nz

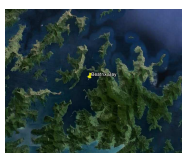
Desktop Summary of Current Level of Science and Understanding of the Cumulative Ecological Impacts of Mussel Farms Ring-Fencing Coastlines such as Beatrix Bay, Marlborough Sounds

Report Status

Version	Date	Status	Approved By:
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V 2	17 May 2014	Rev. 1	
V 3	18 May 2014	Rev. 2	

It is the responsibility of the reader to verify the currency of the version number of this report.

Shaw Mead *BSc, MSc (Hons), PhD*
Tim Haggitt *BSc MSc (Hons), PhD*



Cover page: Satellite image of the complex Marlborough Sounds system of drowned river valleys (Source – Google Earth)

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Executive Summary

In relation to the impacts of mussel farming in the Marlborough Sounds, especially in Beatrix Bay, the Pelorus Boating Club and the Kenepuru and Central Sounds Residents Association commissioned eCoast to develop a desk top summary of the current level of science and understanding of the cumulative ecological impacts of mussel farms ring-fencing coastlines such as Beatrix Bay. The following specific issues were addressed, and below each in this summary is the short answer – details are provided in the body of the report – specifically:

- Whether the 2009 Cawthron Report on the review of ecological effects of aquaculture continues to represent the current position re the understanding of the ecological impact of mussel farming;

YES

- If Section 8 continues to represent the particular information gaps on the ecological impact of mussel farming;

YES

- Whether these information gaps continue to include the impact of mussel farms on indigenous ecosystems in the inter-tidal and sub-tidal areas inshore of marine farm ribbons;

YES

- Whether in our opinion that it is at least likely, if not probable, that the continuous ribbon of mussel farms circumnavigating Beatrix Bay (see map <http://maps.marlborough.govt.nz/viewer/?webmap=6af1f32120314f569f780dafba264> is having/has had a cumulative and potentially serious impact on the indigenous ecosystems inside of them;

YES

- Whether these inter-tidal and sub-tidal areas are important breeding and living grounds for indigenous species, including recreational fish such as blue cod;

UNKNOWN, BUT LIKELY

- Whether the application adequately addresses these issues;

NO, THE APPLICATION DOES NOT ADDRESS THESE ISSUES

- Whether the application addresses the impact that light shading, nutrient depletion and current softening/alteration will have on the reef that the application seeks to surround;

NO, THE APPLICATION DOES NOT ADDRESS THESE ISSUES

- The current knowledge associated with the change in plankton composition brought about by mussel farms and the impact, or unknown impact, of this, and;

NO, THE APPLICATION DOES NOT ADDRESS THIS ISSUE

- The statements in the Marlborough District Council Planners Report at paragraphs (34-35) suggests that from an ecological perspective the cumulative impact of mussel farms is no more than minor because mussels filter less than 1% of the water that flows through farms and consume less than 5% of the plankton from the water passing through the farm.

THIS STATEMENT IN THE MDC PLANNERS REPORT IS HUGEY INCORRECT

Due to the cumulative impacts of aquaculture, it is likely that Beatrix Bay, and potentially many parts of the Marlborough Sounds when activities such as intensive finfish farming are considered, is experiencing death by a thousand cuts, i.e. creeping normality, the way a major change can be accepted as the normal situation if it happens slowly, in unnoticed increments, when it would be regarded as objectionable if it took place in a single step or short period. It is analogous to the landscape amnesia that led to the long-term environmental degradation of Easter Island, which can explain why the natives would, seemingly irrationally, chop down the last tree on the island (Diamond, 2005).

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1 Background

The Pelorus Boating Club and the Kenepuru and Central Sounds Residents Association commissioned eCoast to develop a desk top summary of the current level of science and understanding of the cumulative ecological impact of mussel farms ring-fencing coastlines such as Beatrix Bay. eCoast consultants have had previous experience with evaluating the impacts of mussel farming in Beatrix Bay during the Environment Court hearings associated with the development of large (42.25 ha) mussel farms in the open areas of the Bay in 2001/02.

In particular, the Pelorus Boating Club and Kenepuru and Central Sounds Residents Association has requested that we consider:

- Whether the 2009 Cawthron Report on the review of ecological effects of aquaculture continues to represent the current position re the understanding of the ecological impact of mussel farming;
- If Section 8 continues to represent the particular information gaps on the ecological impact of mussel farming;
- Whether these information gaps continue to include the impact of mussel farms on indigenous ecosystems in the inter-tidal and sub-tidal areas inshore of marine farm ribbons;
- Whether in our opinion that it is at least likely, if not probable, that the continuous ribbon of mussel farms circumnavigating Beatrix Bay (see map http://maps.marlborough.govt.nz/viewer/?webmap=6af1f32120314f569f780da_fba264 is having/has had a cumulative and potentially serious impact on the indigenous ecosystems inside of them;
- Whether these inter-tidal and sub-tidal areas are important breeding and living grounds for indigenous species, including recreational fish such as blue cod;

- Whether the application adequately addresses these issues;
- Whether the application addresses the impact that light shading, nutrient depletion and current softening/alteration will have on the reef that the application seeks to surround;
- The current knowledge associated with the change in plankton composition brought about by mussel farms and the impact, or unknown impact, of this, and;
- The statements in the Marlborough District Council Planners Report at paragraphs (34-35) suggests that from an ecological perspective the cumulative impact of mussel farms is no more than minor because mussels filter less than 1% of the water that flows through farms and consume less than 5% of the plankton from the water passing through the farm.

It is our understanding that the last dot point does not reconcile with what residents are observing and nor to the accepted fact that inside lines on mussel farms grow up to 30% slower than outside lines.

1.1 Previous Reviews and Studies in Beatrix Bay (2001/02)

As mentioned above, we have previously undertaken reviews and studies of Beatrix Bay – 2 reports from those investigations are provided:

- Mead, S. T., K. P. Black and A. Longmore, 2001. *The Sustainability of Marine Farming in Beatrix Bay, Marlborough Sounds*. For the Marlborough Sounds Trust, March 2001.
- Mead, S. T., 2002. *Ecological Survey of Beatrix Bay, Marlborough Sounds*. Report prepared for the Marlborough Sounds Trust, July, 2002.

It is recommended that these reports are read in conjunction with this desktop review, since many of the points focussed on here are investigated in detail. The first report focussed on the following points:

1. Carrying capacity;
2. Benthic impacts;
3. Staging will avoid an impact on other farms, and;
4. Whether or not mussels are good indicators of overall bay ecological health.

The second report provides information about the existing biota that inhabits Beatrix Bay, especially the reef and shallow subtidal communities of the bay margins where comparatively more complex and stable topography is present and therefore higher species diversity would be expected (e.g. Mead and McComb, 2002; Pickering and Whitmarsh, 1996; Pratt, 1994).

It is notable that when these reports (and subsequent Environment Court evidence) were prepared in 2001/02 to consider the concerns of the Marlborough Sounds Trust with respect to the impacts of mussel farms on the wider ecosystem of Beatrix Bay, 160 ha of mussel farms were operating. Since then, new permits have been granted and many farms have been extended seawards by several lines since this time – today a total of 297 ha has been allocated to aquaculture (~15% of the Bay).

2 Site Description

Figure 2.1 provides a map of Beatrix Bay in the Marlborough Sounds, with the fringing mussel farms overlaid.

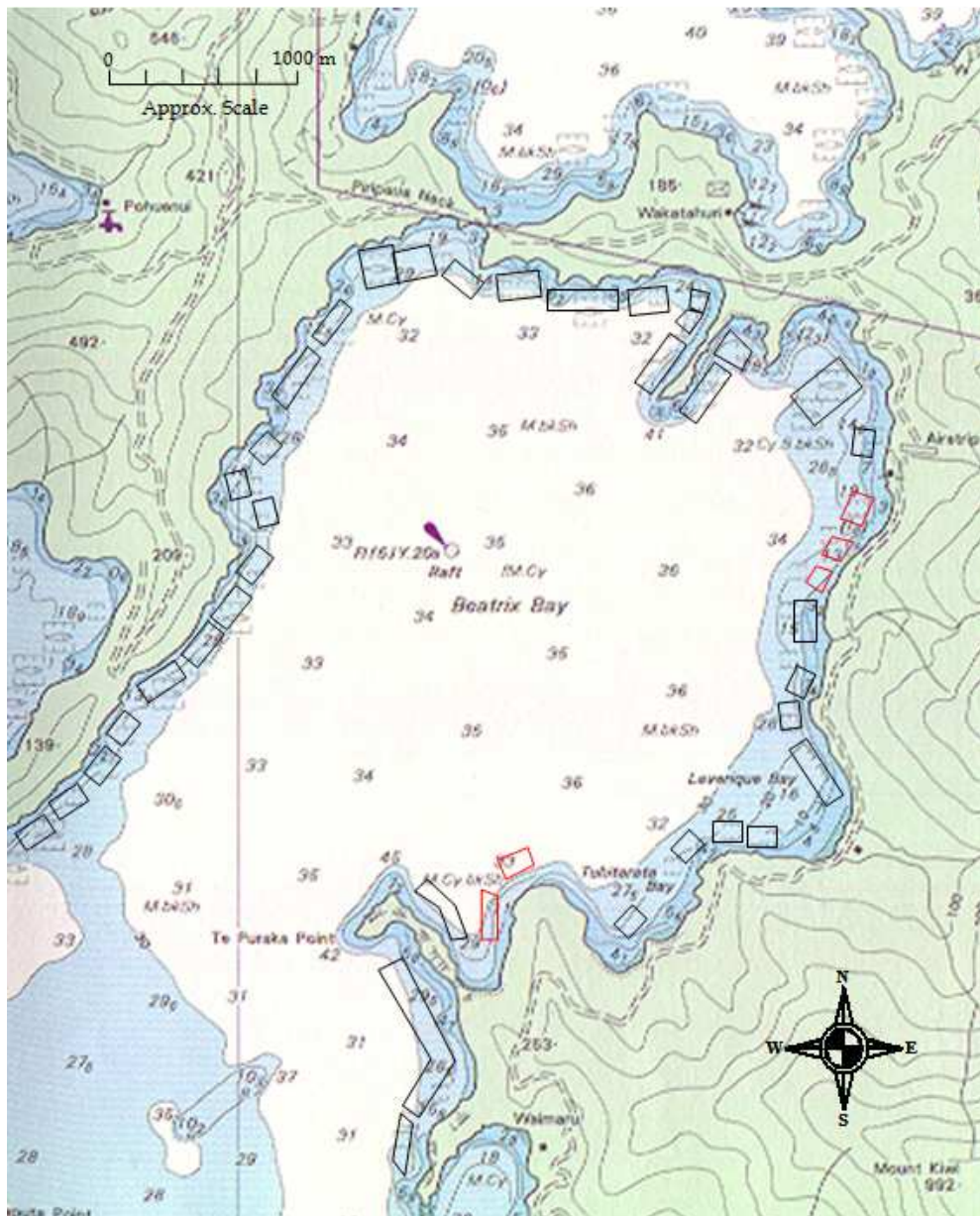


Figure 2.1. Location map of Beatrix Bay and the fringing mussel farms – the red farms have been installed since 2002, while many other farms have been extended seawards since this time (not shown). (Source – Black *et al.*, 2001)

Beatrix Bay is located in the Marlborough Sounds, linked to Cook Strait by the Pelorus Sound, which is a drowned river valley system about 55 km long, with several side arms and bays. Beatrix Bay is one of three water bodies forming one

side arm off Pelorus Sounds (cover image and Figure 2.1). Beatrix Bay is roughly circular (about 4.5 km diameter) and mostly 30-35 m deep (Figure 2.1).

A fundamental aspect of a good mussel farm location is high current flow. High currents provide a large volume of water to filter food from, reduce impacts on the seabed and mix the local chlorophyll levels (Waite, 1989). Regions of slow currents are more likely to incur benthic impacts and receive less food. Beatrix Bay has slow currents and a flushing time of over 2 weeks (i.e. it is poorly flushed), i.e. *Beatrix Bay is fundamentally not an optimum location for mussel farms.*

2.1 Ecological Value of the Marlborough Sounds

The concerns of the Pelorus Boating Club and the Kenepuru and Central Sounds Residents Association are associated with the cumulative and potentially serious impacts of intensive mussel farms on the indigenous ecosystems of the Marlborough Sounds. Therefore it is important to have some background to the ecological value of the Marlborough Sounds, both nationally and internationally.

New Zealand's unique coastal waters with the high number of endemic¹ species (e.g., 60% rock pool fish species, >90% of marine molluscs, and 44% of all breeding seabirds are endemic), which along with the loss of more than 70% of original habitat (terrestrial – marine is unknown), makes New Zealand one of 34 priority global biodiversity hotspots i.e., areas that are very important to global biodiversity and are facing extreme threats. The Convention on Biological Diversity, the associated New Zealand Biodiversity Strategy and parts of the RMA and NZCPS are all directed at maintaining New Zealand's unique biodiversity, with the responsibility for its continued existence entirely ours – it cannot be conserved in nature anywhere else in the world.

Marlborough's extensive coastline is no exception when it comes to a diverse marine environment with habitats ranging from the common-place and typical, through to

¹ Endemic species are those that occur naturally in New Zealand, they evolved or migrated here without any assistance from humans.

significant sites that support rare, unique or special species (Davidson *et al.*, 2011). A range of biophysical factors have contributed to a highly complex marine environment, and this physical complexity has resulted in a unique assemblage of species, habitats and communities. No other coastal area in New Zealand exhibits this enormous range of habitat complexity. Indeed, by applying the New Zealand Marine Environment Classification methodology (ME, 2005), it is likely that the Marlborough Sounds is New Zealand's most biologically diverse marine environment.

The Marlborough Sounds has a large number of important and threatened marine species, as well as endemic species that are found nowhere else in the world, many of which are vulnerable to a wide range of threats (Davidson *et al.*, 2011). Intensive mussel farming is one of these threats, which has a range of impacts on the marine environment, including impacts on the water column, the benthos and, due to the vast numbers of mussels being grown in the Sounds, the carrying capacity of marine ecosystems.

3 Review of Cawthron (2009)

In 2009, the Cawthron institute produced a comprehensive report entitled “Sustainable Aquaculture in New Zealand: Review of the Ecological Effects of Farming Shellfish and Other Non-fish Species” (Cawthron, 2009). The review focused on ecological issues pertaining to seabed-related effects; water-column related effects; and, far-field (wider ecosystem) ecological effects (habitats, fishes, marine mammals, seabeds, pathogens, genetics).

The report also identified key gaps in the general understanding of aquaculture-related effects, with some of the core issues being:

- Limited information on the actual rates of sedimentation occurring beneath and adjacent to marine farms;
- Paucity of information regarding the effects of aquaculture and associated biodeposits on high value reef communities that can be found in close proximity to some farm areas;
- Clear deficiency in information surrounding the effects of marine farms on the wider food web and in particular, wild fish assemblages;
- Limited information concerning the effects of bivalve aquaculture on the composition of plankton communities, which in turn may have wider ecological effects on the food web.

These information gaps are presently still relevant, particularly for Beatrix Bay aquaculture activities in an area of coastline that is intensively farmed. We are particularly critical of the fact that many resource consents do not evaluate or consider the wider carrying capacity of the system they will be occupying in tandem with providing an assessment of likely cumulative environmental effects. Deficiencies of this nature are especially applicable to the currently proposed marine farm, which if established, will essentially result in a complete ring-fence of the

northern end of Beatrix Bay in aquaculture farms, and will impact on a hydrodynamic 'hot-spot' in the bay – cumulative impacts and impacts on the wider environment have not been considered.

Furthermore, we are sympathetic to the notion that carrying capacity modelling approaches (e.g., Jiang and Gibbs 2005) have to be undertaken within Beatrix Bay before any new resource consents are granted. Indeed, concerns about carrying capacity, cumulative impacts and impacts on the wider environment were a concern in Beatrix Bay at the turn of the century, but since then permitted farming area in the bay has risen from 160 ha to 297 ha. Some carrying capacity modelling and ecosystem modelling of Beatrix Bay has been carried out (James, 2000; Ren *et al.*, 2009), however, the findings of these investigations have not been applied to resource consent applications for farming permits, and unfortunately they are shellfish production models that ignore other species in the Beatrix Bay ecosystem that rely on phyto and zooplankton.

In summary, the 2009 Cawthron Report on the review of ecological effects of aquaculture continues to represent the current position re the understanding of the ecological impact of mussel farming, and Section 8 continues to represent the particular information gaps on the ecological impact of mussel farming

4 Potential Cumulative Impacts of Beatrix Bay Mussel Farms

4.1 Cumulative Impacts of Aquaculture

Given the level of aquaculture within Beatrix Bay there are a range of potential cumulative impacts² that could arise. In a recent report, Cornelisen (2013) provides an overview of four types of scenarios that have the potential to lead to cumulative impacts arising from aquaculture developments. These are:

- A. Additive effect of increasing numbers of marine farms;
- B. Additive effect of a single stressor from multiple sources in addition to marine farms;
- C. Additive and synergistic effects of multiple stressors from a single source; and;
- D. Additive and synergistic effects of multiple stressors from multiple sources.

Of these, type A: is likely to be the most pertinent to continuous farms along the perimeter of inlets, such as Beatrix Bay.

Cornelisen (2013) summarises the main effects associated with extractive forms of aquaculture, i.e., mussel and oyster aquaculture, that may lead to cumulative ecological effects on the wider ecosystem, such as oligotrophication (oligotrophic environments offer little in the way of nutrients to sustain life), changes in the abundance and composition of plankton – which may lead to down-stream effects on the food web. It is also suggested that farming of macroalgae could add to the oligotrophic process by removing dissolved nutrients from the water column. This is of concern with respect to the current application which seeks consent to grow *Macrocystis pyrifera*, *Ecklonia radiata*, *Gracilaria*, *Pterocladia lucida* and *Undaria*.

Cumulative impacts stemming from intensive mussel aquaculture have the potential to occur at medium (bay-wide) and large (regional) scales, and will persist providing the level of farming exceeds the natural carrying capacity of the system to maintain

² A cumulative impact/effect is referred to in Section 3 of the RMA (1991) as an *effect which arises over time or in combination with other effects*

bivalve growth and biomass and general functioning of the wider ecosystem (see Dame and Prins 1991).

Is there evidence for ecosystem type carrying capacity effects already manifest in Beatrix Bay? Because there has been limited monitoring studies within Beatrix Bay it is difficult to establish quantitatively changes in natural biota that points directly to surpassed carrying capacity of the system. Cawthron (2009) suggests that culture areas in New Zealand constitute a small fraction of the total area of embayments in New Zealand stating the “heavily farmed” Beatrix Bay has approximately 5% of the bay area under culture, thereafter concluding that bay-wide scale breaches of ecological carrying capacity are unlikely to have occurred from the level of culture within the embayment. However, as there is no monitoring or long-term data relating to native species or detailed carrying capacity models (e.g., Jiang and Gibbs 2005) developed for Beatrix Bay to argue either way, statements of this nature are very misleading, and based on the evidence available even back in 2001 (mussel growth rates and the ‘health’ of the existing ecological communities) is incorrect.

Furthermore, when the hydrodynamics of Beatrix Bay are taken into account (low current flows and poor flushing), the correct percentage of space in the Bay being farmed is applied (according to the Planners report 15% of the total water space is Beatrix Bay is being farmed³, not 5%,) along with the quarter of a billion mussels in the Bay, as well as the often low phytoplankton concentration in the Bay (Ross *et al.*, 1998) and the distribution of the farms in an almost continuous ribbon around the Bay, there is obviously a strong possibility that cumulative impacts could be negatively affecting the inter-tidal and shallow sub-tidal ecosystems inside the farms and the carrying capacity of the bay. Indeed, from the basic carrying capacity modelling presented by James (2000), that did not include components of the wider environment (i.e. a mussel carrying capacity model, how many mussels could the Bay stock without taking into account other species reliant on phytoplankton or impacts on the foodweb), indicated that carrying capacity in the Bay may already have been exceeded – i.e. incorporating an additional 42 ha of farms was predicted

³ With 297 ha with farming permits in the approximately 1,936 ha of Beatrix Bay, this equates to 15.34% of the Bay being utilized.

to lead to an increase of 6 weeks to harvest time. Since then an additional 137 ha of farms have been permitted without considering the carry capacity of the Bay in the application.

The cumulative impacts of mussel farms in the Marlborough Sounds, and in Beatrix Bay, have been considered in a number of investigations, although the findings do not seem to be taken into account when considering new, individual farm applications. For example, Ren *et al.* (2010) applied a generic ecosystem model to Beatrix Bay and found that in an intensive culture embayment in the Pelorus Sound of New Zealand, the model successfully captured main features of the observed system behaviour. The model simulations demonstrated that the mussel cultivation can have considerable effects on the ecosystem of the bay including food depletion and nutrient cycling.

The debate about impacts on carry capacity and the indigenous flora and fauna of the Marlborough Sounds continues because local non-scientists that have and are seeing real changes to the marine environment are not considered in decision making and these observations are dismissed as anecdotal, but also because there has never been any monitoring in place to measure impacts and changes to the native ecology. How can there be no monitoring of an extractive (i.e. phytoplankton/nutrients) and disturbing (the benthos) activity in the marine environment, which is a commons to all New Zealanders, that spans 297 ha and 15% of Beatrix Bay?

4.2 Filtration Rates Reported in the MDC Planner's Report

The statements in the Marlborough District Council Planners Report at paragraphs 34-35 (Section 42A Report for a Coastal Permit application in Beatrix Bay) suggest that from an ecological perspective the cumulative impact of mussel farms is no more than minor because mussels filter less than 1% of the water that flows through farms and consume less than 5% of the plankton from the water passing through the farm (which is based on the current opinion of NIWA from surveys of farms in Beatrix Bay and Port Underwood). However, this does not reconcile to what residents are

observing and nor to the accepted fact that inside lines on mussel farms grow up to 30% slower than outside lines.

The initial concern with this statement in the planner's report is that if the mussels filter <1% of the water passing through the farm, it would be impossible to consume more than <1% of phytoplankton in the water passing through the farm even if it was possible for mussels to extract 100% of plankton from the water column. Therefore, we will consider that the statement is referring to extraction for 5% of the plankton in the water that is filtered by the mussels (i.e. <5% of <1%). Both of these percentages are grossly under-represented when the available information is considered.

Review of the Port Underwood ecological assessment (NIWA, 2012) which includes an estimate of filtration rates does not agree with these numbers from the Planners report, and has little bearing on other sites – indeed, the known filtration rates for Beatrix Bay are far greater than 5%, while extraction of the most palatable phytoplankton (dinoflagellates) by greenshell mussels can be greater than 80% (Hayden, 2006).

NIWA (2012) do not detail the methods used to calculate filtration rates, although they state that by combining mean current speed and direction with the average stocking densities and filtration rates of mussels, the contribution of each existing farm in Port Underwood to phytoplankton depletion can be estimated. From this they estimate that approximately 10-14% of the water flowing through the farms around the perimeter of Port Underwood is filtered, i.e. an order of magnitude higher than stated in the planning report.

When Port Underwood is related to Beatrix Bay, the physical/hydrodynamic differences indicate that it is very unlikely, if not impossible that Beatrix Bay would have similar filtration rates as Port Underwood. Beatrix Bay is more than 2x as deep as Port Underwood, has a flushing period of up to 8x longer than Port Underwood, and has considerably lower currents than Port Underwood (Kuku Mara Partnership, 2000; Ross *et al.*, 1998; Cawthron, 2011; NIWA, 2012). These differences imply that

the percentage of water filtered in Beatrix Bay will be significantly greater than that at Port Underwood, which is supported by existing studies of Beatrix Bay.

Ogilvie *et al.* (1998) found chlorophyll depletion of up to 72% in small farms in Beatrix Bay. Waite (1989) found that at current speeds of 2-10 cm.s⁻¹, *P. canaliculus* consumed 15-60% of available food in the water column – Beatrix Bay current speeds are at the lower end of these current speeds (Kuku Mara Partnership, 2000; Cawthron, 2011), which results in higher extraction percentages. Mead *et al.* (2001) concluded:

*“While the consistent occurrence of “adequate food” is probably the single most import factor determining the suitability of the Marlborough area for mussel culture, food sources vary between embayments and within mussel farms. It is our opinion that the concurrence of slow current speed and high stock density will accentuate the depletion of food resources within the mussel farm. **The currents in West Beatrix Bay are very small and so at least 15-60% of food could be consumed within the farm.** West Beatrix Bay is subject to low current flows and therefore limited food supplies.”*

If we considering the number of mussels in Beatrix Bay (250,000,000 – Cawthron, 2011), filtration rates of Greenshell mussels (38 l/day for large adults – Hayden, 2006) and the volume of the Bay (~658.24 million litres), and reduce the filtration rate to 19 l/day to allow for all size classes of mussel, the farms in Beatrix Bay are potentially filtering 4.75 billion litres of seawater, or 7.2x the volume of Beatrix Bay. Obviously we have current speeds and tidal exchange to consider, and mussels do not extract 100% of the phytoplankton in the water column (although rates can be >80% (Hayden, 2006)). Even so, the known extraction rates in Beatrix Bay and in other areas of the Sounds, and the massive filtration capacity of mussels in Beatrix Bay do not support the statements in the officers report, filtration rates are known to be far in excess of 1%, and phytoplankton extraction is known to be far in excess of 5% of the water passing through farms in Beatrix Bay. Indeed, the available evidence indicates that there are likely to be negative impacts on both mussel farms

(inside lines and farms deeper into the Bay) and the indigenous filter-feeders around the fringes of the Bay.

4.3 Changes in Phytoplankton Composition

In the assessment of effects of water column impacts, Cawthron (2013) go on to promote the idea that during times of limited nutrients (and hence limited phytoplankton growth) the higher levels of dissolved organic nitrogen measured within farms (Ogilvie *et al.*, 2000) is a positive situation for the farm and potentially for downstream sites if the farm is in a higher current location. This suggests that mussel farms are some kind of perpetual-motion engine, because they are able to locally increase phytoplankton through excretion and provide nutrition for themselves and other farms close by. However, there are a range of studies that indicate that it is unlikely to be an indication of a healthy system:

- a) the higher chlorophyll a within the farms is due to excretion from mussels because they are starving, actually feeding on their own body mass to survive due to the low levels palatable phytoplankton available (and according to James (2000) “if mussels are doing well, other biota should also be doing well” – i.e. the contrary situation during these periods of low nutrients, organisms that survive by filter-feeding will be stressed bay-wide), and;
- b) it is well known that different species of phytoplankton have different nutritional value (Grant and Bacher, 1998; Hayden, 2006), some may even inhibit shellfish filtration (Prins *et al.*, 1994), and that the fastest-growing phytoplankton (diatoms) that respond to the nitrogen excreted within stressed mussel farms is likely the least palatable to mussels (e.g. Hayden, 2006). It is food quality that is more important than quantity in accurately predicting growth, and food quality is dependent on species (Grant and Bacher, 1998: Campbell and Newell, 1998).

While the cause(s) of the poor growth is also related to natural oceanographic processes (Zeldis *et al.*, 2008), there is no reason to believe the extra food consumption by continued expansion of mussel farms in Beatrix Bay will not

increase the pressure on existing mussel farms and the other marine life in the Bay. Mussel growth in Beatrix Bay was considered limited by nitrogen availability in the late 1990's (Gibbs and Vant, 1997). Farmed areas have almost doubled since 2000, at which time the Beatrix Bay ecosystem has been considered unhealthy for considerable periods (James, 2000). Through the late 1990's, there was a rapid decline of the condition of the mussels grown in the Pelorus Sound area, and it was suggested that this is a possible indicator of a sustainable production problem (Ross *et al.*, 1999). Re-evaluation of the industry growth data (presented in Mead *et al.*, 2001) should be undertaken to consider the impacts of increased stocking levels in Beatrix Bay on mussel growth rates since 2001, which is also an indicator of how the marine life in the Bay is being impacted. Sustainable development does not occur at the expense of the natural environment and the organisms inhabiting it.

4.4 Cumulative impacts on inter-tidal and sub-tidal areas are important breeding and living grounds for indigenous species

While Information regarding the extent and biological character of Intertidal and subtidal regions in Beatrix Bay have been considered in a broad-scale reviews (Mead, 2002a) and in one-off, often spatially-explicit, studies for resource consent applications, the functions and services that existing habitats play and their role in determining the environmental integrity (trophic-linkages) within Beatrix Bay are not well understood. Based on available ecological inventories it is clear that soft sediment habitat is the predominant habitat type with both intertidal and subtidal rocky reef limited in spatial extent. Nevertheless we contend that due to the paucity of rocky reef habitat, by default it should be regarded as an ecologically significant marine habitat within Beatrix Bay and less weight given to comparisons with other locations within the Marlborough Sound or elsewhere in New Zealand, i.e., because an area has a low to moderate diversity index does not by default make it ecological insignificant. Discretion must be given when evaluating presented measures of ecological diversity (taxa richness, Shannon-Weaver diversity measures, etc.) as for the current situation they represent a snap-shot in time taken from an already disturbed modified/environment and will be context-dependant.

Considering the continuous ribbon of aquaculture farms within Beatrix Bay it is apparent that the majority of farms partly occupy the shallow to mid-depth subtidal regions (5-20m depth) which are generally characterised by the greatest benthic habitat heterogeneity (rocky reef, mud, sand, shell hash, etc.) and greatest biological diversity of the entire bay (Mead, 2002a). Very little is known about how rocky reef habitats and associated biological communities may have changed (or not have changed) within Beatrix Bay since the advent of aquaculture in 1980, so merely stating that a habitat or suite of habitats has low to moderate diversity particularly when viewed in isolation (rather than bay-wide) fails to consider the full ecological functionality (diversity, foraging areas, recruitment areas, habitat linkages) that habitats of this nature may play in the Beatrix Bay system.

The ecological assessment of the current application identifies bedrock, boulder, and cobble reef, sand and shell-hash, mud and shell-hash and mud as the main habitat classes, the latter being the most dominant habitat. Numerous native taxa were observed on and nearby the rocky habitat immediately inshore of the proposed marine farm including low-lying brown algae (thought to be *Stictosiphonia* sp⁴), encrusting coralline algae, and encrusting sponges with fish fauna represented by spotties, blue cod, and butterfly perch, mobile invertebrates represented by kina, 11-armed sea-stars, and cushion stars with heart urchins, brittle stars, sea-stars and scallops found on soft sediment habitats. Likewise, Mead (2002a) in his spatially boarder study lists a comparable rocky reef and soft sediment species matrix. Intertidal regions surrounding the proposed marine farm were not sampled; however, Mead (2002a) suggests that intertidal habitats are typically comprised of cobbles and small boulders that support macroalgae, bivalves, crustaceans, and assorted sessile invertebrates.

Arguably due to the occurrence of species associated with rocky reef and soft sediment habitats within Beatrix Bay (including the current application), it is to be expected that the matrix of habitats are important living and breeding areas for indigenous species, including those species with significant cultural and recreational value (e.g., kina and blue cod). In particular, diet-related studies of fish such as blue

⁴ Note: this species is now placed in the genus *Bostrychia* – Nelson (2013)

cod (Jiang and Carbines, 2002) and spotty (Jones 1988, Rilov and Schiel, 2006) support the significance of Beatrix Bay habitats in fulfilling foraging requirements and are equally likely to fulfil demographic (reproduction, settlement and growth) requirements of the many species encountered. Equally, the occurrence of sandy and shelly bottoms that provide structure and protection which occur within Beatrix Bay (Mead, 2002a) are likely important as nursery and foraging areas for blue cod, although spawning for this species typically occurs in coastal and outer continental shelf waters from late winter to early summer (Davidson *et al.*, 2012). The occurrence of rocky reef and shell-hash habitats are also important for spotty and butterfly perch recruitment (Francis, 1996).

In an aquaculture review, Stimenstad and Fresh (1995) state that growth and survival of animals in estuaries and embayments not only depends on specific habitats but on linkages between habitats and areas within the estuary, which can be altered over the long-term by sustained disturbances such as intensive aquaculture. Therefore, ensuring that the environmental integrity of key habitats is preserved is also paramount particularly in terms of maintaining trophic-linkages, for example intertidal habitats have been shown to be important in terms of foraging and diet for species such as spotty (Rilov and Schiel, 2006).

4.4.1 Cockle Population Dynamics in Laverique Bay, Beatrix Bay (From Mead, 2002b)

The majority of marine species that inhabit the intertidal and shallow sub-tidal zones around Beatrix Bay have a planktonic early life history, i.e. they have a larval stage where they exist as zooplankton before settling in/on suitable substrate. International studies have uncovered some of the potential effects of extensive shellfish culture that have not been considered for Beatrix Bay or in the application.

For example, in the past mussels have generally been viewed as specialised herbivores. However, studies dating back to 1933 (Nelson – cited Davenport *et al.*, 2000), have appreciated that bivalves ingest zooplankton. More recently a series of studies headed by J. Davenport at University College Cork in Ireland have demonstrated that blue mussels ingest zooplankton up to 6 mm long, as well as bind them in mucus and expel them as pseudo-faeces (Davenport *et al.*, 2000; Lehané

and Davenport, 2002). These zooplankton species have been identified as mollusc eggs and larvae, polychaete larvae and small adults, bryozoan larvae, crustaceans (copepods, crabs, barnacles, amphipods, ostracods), echinoderm larvae and fish eggs. These findings have direct implications for mussel culture that will have direct impacts on local recruitment of benthic animals and pelagic fish, as well as increasing competition for primary production resources in the areas they are located (Davenport *et al.*, 2000), since zooplankton are a major link to higher trophic levels. With a near continuous ribbon of mussel farms between the open water and the preferred habitats of many benthic species (i.e. the intertidal and shallow sub-tidal areas), there is obviously potential to significantly impact the larval population in Beatrix Bay.

Little attention has been paid to the effect of mussels on zooplankton populations in Beatrix Bay through competition for phytoplankton resources (Ross and James, 1996; Zeldis *et al.*, 2004). Davenport's work in Ireland has prompted concerns over the volumes and types of zooplankton that cultured mussels in the Marlborough Sounds could consume (Wilson, 2002). The results of these investigations are presented in Zeldis *et al.*, (2004), which found that mussel gut contents had numerous copepod parts, copepods and larval bivalves present. This is definitely an important factor in terms of ecosystem impacts that have not been considered since, in addition to the diverse planktonic algal species and zooplankton, many marine organisms spend a part of their early life history as planktonic larvae (e.g. crabs, rock lobster, bivalve molluscs, fishes, jelly fishes, echinoids, polychaetes, etc.). Indeed, the zooplankton consumption estimates of Davenport and those made from seston trawls undertaken in the Marlborough Sounds indicate very large numbers of zooplankton can be consumed by cultured mussels which will have direct impacts on local recruitment of benthic animals and pelagic fish (nearly all New Zealand coastal fish species have pelagic eggs (Cole, 2002)), as well as increasing competition for primary production resources in Beatrix Bay. However, like many areas of knowledge with respect to the impacts of shellfish farming through the ingestion of zooplankton, the ecological significance and contribution to mussel energetics remain questionable (Ren *et al.*, 2010), i.e. mussels are ingesting zooplankton (Zeldis *et al.*,

2004), but what fraction of the total and how this impacts on the wider environment is unknown.

Possible evidence of increasing competition for primary production resources has previously been presented by Grange (1997). Grange carried out surveys of natural cockle populations in Croisilles Harbour and Delaware Inlet when NIWA was approached to provide advice on whether granting additional marine farm consent in Oyster Bay (Croisilles) (Figure 4.1) may result in exceeding the carrying capacity in the bay and thus adversely affect natural marine communities, especially filter feeders that may compete with farmed mussels and oysters in the bay (Grange, 1997).



Figure 4.1. Location map of Oyster Bay in Croisilles Harbour.

The reasons for this concern centred around largely *anecdotal* evidence that farmed mussels in the Bay were taking longer to reach peak condition in recent years than previously, and that the cockle populations at the head of the bay had declined in recent years. Grange found differences in the population structure of the Oyster Bay cockles (mussel farms) in comparison to the Delaware Inlet cockles (no mussel farms) that prompted him to conclude that the study provided data which may give the very first glimpse that native species are being compromised and that the

potential for aquaculture within the existing licenses may already be sufficient to exceed carrying capacity, to the detriment of both the aquaculture industry and native species (Grange, 1997). In Grange's opinion, the granting of additional consents may not be wise until further research had been undertaken.

The anecdotal evidence in relation to changes in the shoreline species at Beatrix Bay shows close similarities with the Oyster Bay case, i.e. mussels taking longer to grow and reach peak condition and decline in natural shellfish populations. While there are several deficiencies to the Grange study, since it is a snapshot in time with no links to nutrient and phytoplankton levels, no previous studies of natural shellfish populations, etc., Grange's conclusions are based on the relative abundances and population structures in areas where there are marine farms and where there are none.

Prompted by Grange's work, a similar study was undertaken in Laverique Bay that is located on the south eastern side of Beatrix Bay (Figure 2.1). Similar to Oyster bay, Laverique Bay once provided high densities of cockles and pipis, and the anecdotal evidence suggests that the numbers of these shellfish (along with feral green mussels, seaweed and kina) declined over the past decade (1992-2002). The results of 27 x 0.1 m² quadrats sampled down transects between the high and low tide range on 6 October 2002, show a bimodal population structure, with a dearth of medium sized cockles indicating high mortality in this age class, very similar to that found at Oyster Bay by Grange (Figure 4.2). As can be seen from this series of graphs, the Delaware Bay population in an area absent of extensive mussel farms has a 'normal' bell-shaped population structure, while both Oyster Bay (Croisilles) and Laverique Bay (Beatrix) have few individuals in the mid-age classes, suggesting high mortality is occurring (Grange, 1997). The presence of individuals in the lower size classes indicates that settlement of cockles is still occurring (Grange, 1997).

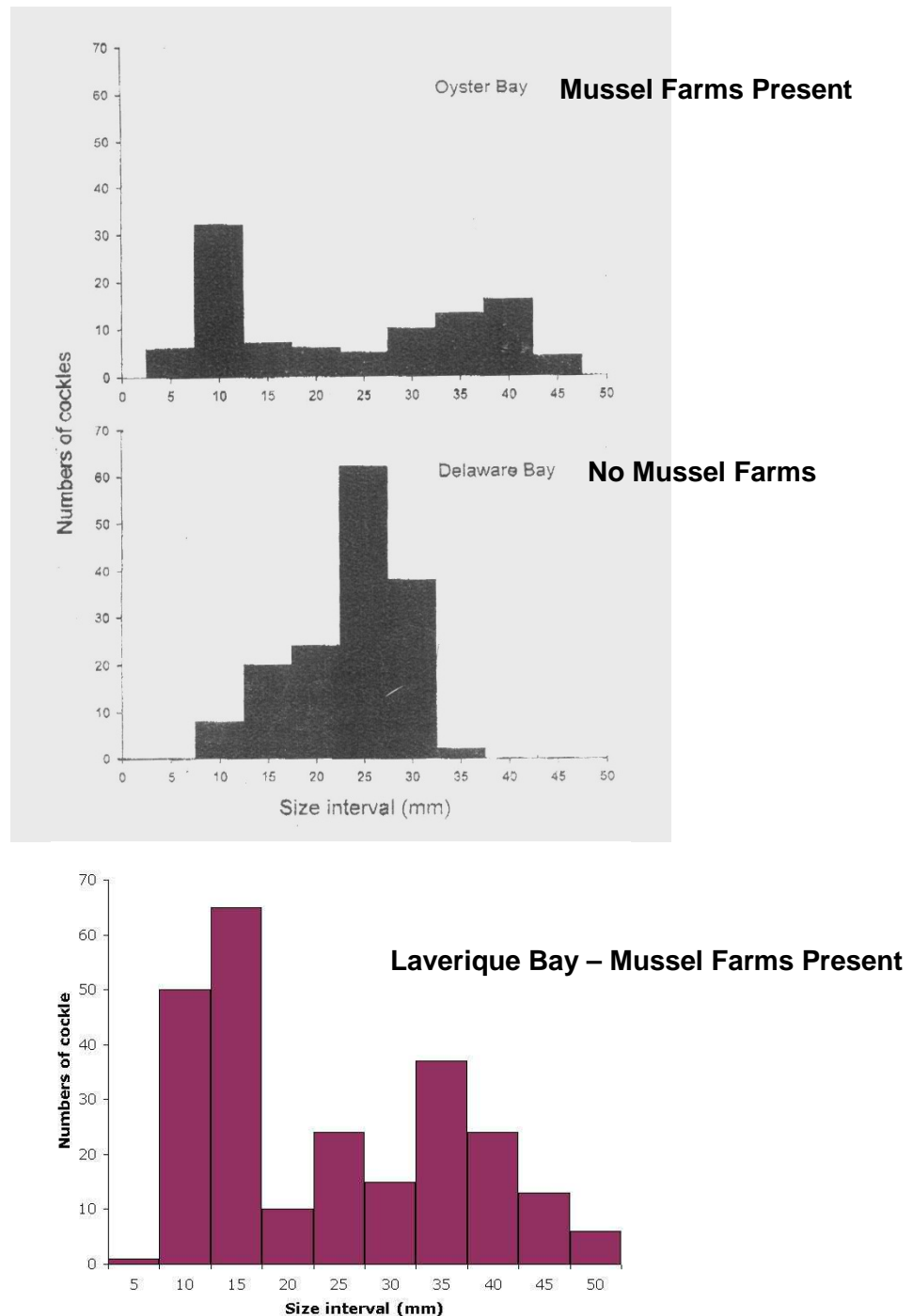


Figure 4.2. Population structure of cockles at Oyster Bay (top), Delaware Bay (Middle) and Laverique Bay in Beatrix Bay (bottom).

Unlike Oyster Bay, the long-term chlorophyll *a* and nutrient monitoring in Beatrix Bay shows that there has not been significant decreases in past years (not with the duration that could impact on population structure in this way since the large individuals are likely to be 10-15 years of age (Grange, 1997)), which was put forward by Grange as a way of conclusively showing whether this was the reason for

decline. It is difficult to ascertain whether this is a local effect (e.g. due to nearby farms), a bay wide condition, or a state of a much larger area that is linked to climatic variables. Even so, the same conclusions and advice advocated by Grange are warranted, i.e. it is likely that native species are being compromised to the detriment of both the aquaculture industry and native species. A precautionary approach should be taken and the granting of additional consents may not be wise until further research into the impacts of mussel farms on the wider Beatrix Bay ecosystem have been undertaken – that was the case in 2002, and it is still the case today.

As the biomass of bivalves increases, as it has/is in Beatrix Bay through the continual addition of mussel farms, the matter and energy necessary to maintain these animals increases proportionally at a greater rate (Dame and Prins, 1998). This will have direct effects on the existing ecosystem e.g. extraction of phytoplankton and zooplankton (feeding and pseudo-faeces), extraction/loss/changes of nutrients (harvesting and changes to nutrient cycling and dominant nitrogen source), habitat loss and modification (under farms), etc. In Beatrix Bay these effects may be bay-wide (e.g. nutrient limitations), although magnified further into the bay due to the low current (i.e. poor circulation and flushing, re-circulating eddy in the north western corner) and under mussel farms (e.g. loss/change of nutrient cycles and habitats), or limited to species with particular feeding behavior competing for phytoplankton (e.g. zooplankton, tubeworms, bivalves, brachipods, some crustaceans), mid-water feeders competing for zooplankton (kahawai, mullet, wrasse) or bottom feeders (skate, rays, pig fish, stargazer, witch flounder, mullet) and infauna (e.g. heart urchins, brittle stars, bivalves, worms) competing for space. These effects may then impact further up the food web to higher order predators such as kingfish, john dory, witch flounder, birds and cetaceans (Figure 4.3).

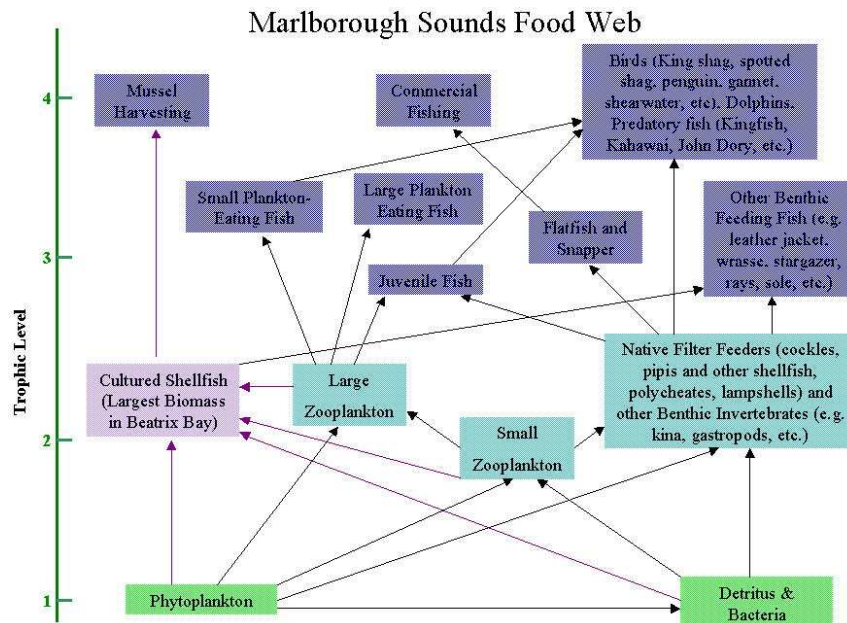


Figure 4.3. Marlborough Sounds marine food web (adapted from Bradford-Grieve, 2002). Note the range of food linkages between cultured mussels and the lower trophic levels.

4.5 Conclusion

These concerns and issues with the impacts of mussel farming in the Marlborough Sounds were highlighted more than a decade ago (e.g. Mead *et al.*, 2001), and there have been many investigations since then that have also indicated the negative and cumulative impacts on the wider environment (e.g. Ren *et al.*, 2010). However, the consenting authority still does not ensure that applicants are considering the wider environmental and cumulative impacts of mussel farming, and the science providers undertaking ecological assessments continue to ignore these issues.

Due to the cumulative impacts of aquaculture, it is likely that Beatrix Bay, and potentially many parts of the Marlborough Sounds when activities such as intensive finfish farming are considered, is experiencing death by a thousand cuts, i.e. creeping normality, the way a major change can be accepted as the normal situation if it happens slowly, in unnoticed increments, when it would be regarded as objectionable if it took place in a single step or short period. It is analogous to the landscape amnesia that led to the long-term environmental degradation of Easter Island, which can explain why the natives would, seemingly irrationally, chop down the last tree on the island (Diamond, 2005).

5 Suitability of the Application to Address Impacts

5.1 Does the application address these impacts cumulative and indigenous?

The application does not adequately address cumulative impacts. It is acknowledged that mussel farming is of great importance to New Zealand's export trade. In turn, the mussel industry is vitally dependent on a high quality marine environment, as fostered by the 'Purpose and Principles' of the Resource Management Act (1991) which are to promote the sustainable management of natural and physical resources (S5(1)) and "safeguarding the life-supporting capacity of air, water, soil and ecosystems" (Section 5(2)a, RMA (1991)).

Considering the impacts of a single farm on the immediate environment does not address the cumulative impacts on the marine environment. Concerns with carrying capacity and impacts on the wider environment in Beatrix Bay are more than a decade old – the Kuku Mara (2000) application developed a carrying capacity model to consider the impacts of additional 42.25 ha farms on either side of the Bay. However, since that time around 140 ha of addition farming space has been allocated in the Bay and the MDC and the science providers have not taken into account the increasing cumulative impacts that this has led to. This is becoming a tragedy of the commons, or a case of 'the straw that broke the camel's back'.

Cawthron state that "*Potential wider ecological impacts considered were the effects of the proposed farm on sea birds, demersal fish and marine mammals.*" and thus considers only the local physical impacts of the presence of the farm on a few mobile species, but do not consider wider ecological impacts, i.e. impacts on ecosystem function of the Bay. Mobile species simply move away if the conditions are unsuitable, but sessile species cannot and so can potentially be impacted, especially if located inshore of the ribbon of farms, as has been observed.

The lack of monitoring of anything in the Bay but mussel growth means that there is very little data to demonstrate the decline in indigenous species observed by local people. Cawthron (2013) cite that the Bay is no longer considered a spawning

ground for Lemon sole, which is evidence suggesting a decline in the health of the bay.

5.2 Does the application address the impact that current softening/alteration?

No, the application does not address the impacts of the proposed farm on water currents, and only partially considers the impacts of the currents on farm impacts (i.e. deposition and water column impacts). Given the location of the farm at the southern end of the point, the influence that this feature has on tidal currents and the influence that mussel farms have on current speeds and directions (slows and modifies, respectively), there is potential to have impacts on the hydrodynamics of a large area of the bay.

Figure 5.1 clearly shows the local current anomaly in the area of the proposed marine farm, but the application does not address how the known impacts of dampening and directional changes to currents will impact on either the nearshore biota inshore of the proposed farm or how modifications to the hydrodynamics at this location will impact on the hydrodynamics of the Bay (e.g. will the reduction of currents lead to smothering of the nearshore rocky reef and cobble communities? Will the changes to the currents at this location have flow-on impacts to other parts of the Bay or farms?).

The application points out that there are other farms at the end of promontories in the Sounds in a 'wrap around' style to indicate that it is therefore acceptable. However, this promontory has an obvious impact on tidal currents and circulation of the head of the bay (Figure 5.1), and so the presence of an 8.982 ha 'wrap around' style mussel farm will impact on Bay hydrodynamics (Figure 5.2), and so on the wider ecology of the Bay. This has not been addressed at all in the application.

The magnitude of impacts to currents due to mussel farms has been quantified and can be significant. Gibbs *et al.* (1991), Boyd and Heaman (1998) and Karayucel and Karayucel (1998) observed reductions in flow through farms of up to 70% compared to currents outside the farms. In addition, longlines have been found to be relatively

impermeable to currents and effectively deflect currents to run parallel to them (Waite, 1989).

Cawthron (2013) conclude that because the currents are strong, pseudo-faeces/biodeposits will not accumulate under the farm wrapping around the end of the promitory, or settle on nearby rocky reef or cobble habitat. However, this conclusion does not take into account reductions of flows of up to 70% (will this lead to deposition on rocky reef or cobble habitat in the 'shadow' of the wrap-around farm?), or the fate of the pseudo- faeces/biodeposits, the latter of which should also be considered in water column impacts (where will this material settle, what is its fate?).

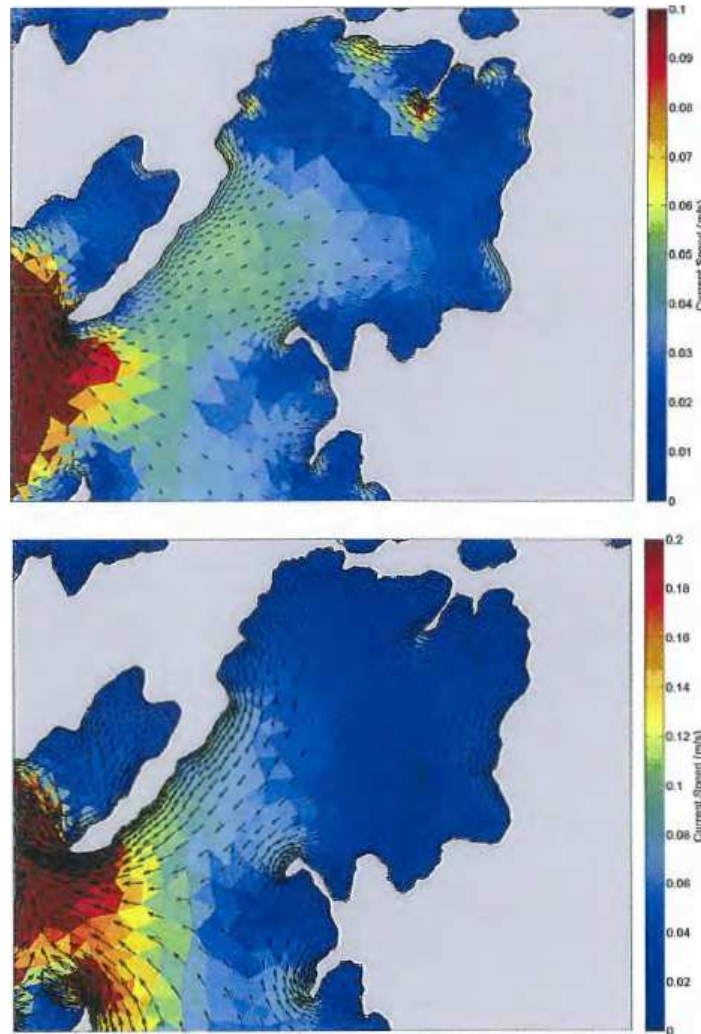


Figure 5.1. Uncalibrated hydrodynamic modelling of Beatrix Bay tides – incoming (upper), outgoing (lower) (Cawthron, 2011). Note the influence of the northern spur in the area of the proposed mussel farm.



Figure 5.2. Schematic of the proposed ‘wrap-around’ mussel farm.

5.3 Does the application address the impact that light shading will have on the reef that the application seeks to surround?

Mussel longlines have the potential to reduce light both in the water column and seabed proper through increased shading from overlying culture structures (longlines) (Inglis *et al.* 2001; McKindsey *et al.* 2011). Given the size and proximity of the current application to subtidal rocky reef habitat, the application clearly fails to adequately address any effects of shading that may occur due to its spatial coverage alone and in tandem with the existing aquaculture farms immediately adjacent. Shading effects must be considered in order to appropriately assess ecological effects associated because it is a persistent effect, i.e., has the potential to occur 365 days a year).

In terms of adverse environmental effects, reduced light via direct shading may lead to a change in the abundance and biomass and species compositions of both benthic microalgae and macroalgae (McKindsey *et al.*, 2011) although effects will depend on the species present. The main macroalgal species occurring on the rocky reef habitat immediately adjacent the proposed marine farm is *Stictosiphonia* sp, which may be negatively impacted by shading, as this species commonly forms distinct zones in the upper intertidal (high light) (Nelson, 2013). Shading effects are not only restricted to algae and can also impact on sessile invertebrate assemblages through alteration in community structure (Gladsby, 1999).

Given that shading effects have not been addressed for the current application and have not been addressed in the context of adjacent marine farms, these shading effects must be evaluated in terms of impacts to the benthos and in particular the inshore rocky reef habitats.

5.4 Does the application consider nutrient depletion?

The application does not consider nutrient depletion, rather it is more concerned with how well the mussels will grow in relatively higher currents in this position in Beatrix Bay. The application mentions that with more mussel farms in an area, a larger amount of material is filtered out of the water column, and goes onto say that there is some anecdotal evidence of decreased mussel growth rates in certain isolated parts of Beatrix Bay, which has been loosely attributed to the number of farms consented in the area. The application then goes on to dismiss this citing it is only farms in constricted parts of the Beatrix Bay that are experiencing these impacts. This is an interesting conclusion given the existing industry data on growth rates (Mead *et al.*, 2001), the modelling of the Beatrix Bay system (Ren *et al.*, 2009), as well as the lack of constrictions in the circular bay (Figure 2.1). The application then goes onto suggesting that higher chloro a inside farms during periods of low nutrients when mussels are wasting away and consuming their own body mass is a positive thing (discussed above).

In summary, the application fails to consider cumulative impacts/effects as referred to in Section 3 of the RMA (1991) i.e., an “*effect which arises over time or in combination with other effects*”.

5.5 Recommendations for Future Investigations:

1. The review of industry data for Beatrix Bay presented in Mead *et al.*, (2001) should be updated to present day to determine the trends in harvest rates (kgs/m of longline seeded). The extent of mussel farms in Beatrix Bay has increased extensively since 2001 (from 160 ha to 297 ha), and even though it is acknowledged that growth rates also vary depending on nutrient inputs into the Marlborough Sounds system, there was a downwards trend in mussel yield and estimates at the time indicated that with the addition of 42.25 ha of mussel farm in the middle of Beatrix Bay, growth rates would decrease bay-wide (i.e. it would take 6 weeks longer to harvest time). More than 3x this area has been added to the Bay since 2001.
2. The literature pertaining to shellfish farming cited in Mead *et al.*, (2001) should be updated – there were already a great deal of concerns about the impacts of mussel farming in Beatrix Bay and worldwide at this time; more than a decade on a range of further investigations have been published.
3. Re-survey the same sites surveyed around Beatrix Bay in May 2002, including the cockle population structure sampling of Laverique Bay and compare against the initial survey (Mead, 2002). While it is acknowledged that ecosystems and communities vary through time due to natural influences, the initial surveys considered a number of sites and a range of habitats which will provide a broad comparison of changes to the Beatrix Bay ecosystem over the past 12 years.
4. This brief desktop review of the points raised by the Pelorus Boating Club and the Kenepuru and Central Sounds Residents Association does not address all of the potential cumulative impacts (e.g. the benthic impacts of 15% of the Bay and associated impacts on nutrient recycling). It is recommended that a review of the other potential impacts (such as those considered in Mead *et al.*, 2001) is also undertaken and updated.

5. Bay-wide monitoring should be undertaken. In many ways the horse has already bolted with respect to monitoring, with 15% of the Bay already being farmed and the reductions in mussel growth rates and occurrence of indigenous species being observed (but not quantified) since the early 1990's. Even so, establishing a Baywide monitoring programme, or indeed a Pelorus Sound-wide monitoring programme will still be of great value for managing the existing system, and considering farming impacts and future applications.
6. Cumulative impacts must be considered during the evaluation of applications for farming permits. It is recognised that previous applications for farming permits in Beatrix Bay have mostly not considered cumulative impacts and impacts on the wider environment, even though these issues have been raised since early this century. Because the consenting body is not considering cumulative and wider environmental impacts, it is not fulfilling its role as environmental managers as defined by the 'Purpose and Principles' of the Resource Management Act (1991) which are to promote the sustainable management of natural and physical resources (S5(1)) and "safeguarding the life-supporting capacity of air, water, soil and ecosystems" (Section 5(2)a, RMA (1991)).

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Attachment 3

**STATEMENT OF EVIDENCE
DR. S. T. MEAD – 10 OCTOBER 2002**

STATEMENT OF EVIDENCE

STATEMENT OF EVIDENCE OF DR. S. T. MEAD – 10 OCTOBER 2002

1.0 Qualifications and Experience

- 1.1 My name is Shaw Mead. I hold BSc and MSc (Hons) degrees from the University of Auckland (School of Biological Sciences), and a PhD degree from the University of Waikato (Earth Sciences). I am currently an environmental scientist and Director at ASR Ltd, which is a marine consulting and research organization. I have 8 years experience in marine research and consulting, have published 9 papers in peer-reviewed scientific journals, and jointly produced over 40 technical reports pertaining to marine ecology, coastal oceanography and aquaculture. I have undertaken hundreds of research SCUBA dives around the coast of New Zealand. I am affiliated to the New Zealand Marine Science Society and the New Zealand Coastal Society (IPENZ).
- 1.2 I have a background in coastal oceanography, marine ecology and aquaculture. I studied for my MSc degree at the University of Auckland's Leigh Marine Laboratory, undertaking subtidal research there from 1994 to 1996 directed at the fertilization success of sea urchins as a basis for the sustainable management and development of the commercial market. The marine ecological components of my Doctorate were directed towards subtidal habitat enhancement of marine structures, while the physical oceanography component was focussed on understanding the effects of coastal bathymetry on wave breaking characteristics using field measures and hydrodynamic numerical modelling. More recently, I have been involved in assessments of physical and ecological effects of marine construction, oil industry and aquaculture ventures.

2.0 Scope of Evidence

- 2.1 In preparation for this evidence I have reviewed a large volume of scientific literature, unpublished theses and CDROM databases and technical reports pertinent to shellfish aquaculture, mussel industry data, the existing information and expert evidence supplied by Kuku Mara for several resource consent applications and appeals in the Marlborough Sounds area, including studies undertaken in the Marlborough Sounds, those specific to the Beatrix Bay carrying capacity model (journal papers, popular articles, technical reports, etc.), expert evidence for similar projects in the Marlborough Sounds and New Zealand, and international work relevant to the issues of water quality and carrying capacity in Beatrix Bay. In addition, in May this year I undertook a qualitative ecological survey of Beatrix Bay comprising SCUBA dives (diver transects recorded on underwater video) and remote-video records (SM1) and supervised a shellfish survey in Laverique Bay (south eastern Beatrix – SM1) to gain a better understanding of the ecology of Beatrix Bay and the impacts of existing marine farms in Beatrix Bay.
- 2.2 My evidence concerns the impacts of the proposed West Beatrix Bay marine farm on the Beatrix Bay environment and on the existing farms. Although there is little available data on the impacts of marine farming on the natural environment in the Marlborough Sounds, since the vast majority of work has been directed at the mussel farming industry, some studies do exist. In addition, there are several international studies that provide insight into the likely impacts of extensive mussel farming.
- 2.3 I will describe some of these studies that are relevant to the addition of a 42.5 ha mussel farm in north western Beatrix Bay, consider the results of the modelling and the application of these results to the health of the wider environment in relation to the definition of sustainable carrying capacity, and detail my concerns with the applicability of the methodology proposed for assessing impacts of and

managing mussel farms in order to further clarify my opinion that the Beatrix Bay ecosystem is already stressed and that to add a large scale marine farm, especially to the north western part of the Bay, would have significant impacts on its health. Indeed, after reviewing the work carried out by the many different scientists in Beatrix Bay since the mid-1990's, it is clear to me that many are of a similar opinion, that West Beatrix Bay is already under stress and not a good place to locate a 42.5 ha marine farm.

Impacts of Mussel Farming

2.4 Mussel farms can have a wide range of impacts on the marine environment on which it depends through habitat modification and the lowering of water quality (Kaspar *et al.*, 1985; Hatcher *et al.* 1994; Barranguet 1997). In an area such as Beatrix Bay, that has a relatively long water residence time (Gibbs *et al.*, 1992), low current speeds (James, 2000), is nutrient limited (Ross *et al.*, 1999) and where previously feral populations of mussels were habitat restricted to the hardshores of the bay, development of extensive mussel farming has greatly increased the biomass of this species to far higher levels than previously existed, making cultured mussels the keystone species in the Beatrix Bay ecosystem. As a biologically active keystone species, cultured mussels in Beatrix Bay have a large impact on the functioning of the ecosystem. Figure SM2 is derived from the Food and Agriculture Organisation of the United Nations "Guidelines for the Promotion of Environmental Management of Coastal Aquaculture Development" (Barg, 1992). This diagram is a summary of the possible ecological effects of mussel farming. Evidence presented by previous witnesses has already addressed many of these impacts in detail, and so this will not be repeated in detail here. However, it should be noted that there are many more potential impacts on the marine environment than have been addressed or discussed in the Assessment of Environmental Effects that supports the application for the West Beatrix Bay marine farm. In addition, the majority of these impacts have the potential to negatively affect the Beatrix Bay ecosystem to differing degrees. It is

true and has often been stated, that the marine ecosystem and the interactions such as those depicted by the food-web (SM3) are very complex, making it difficult to assess how the direct effects of mussel farming impact on the wider environment. Even so, these difficulties do not warrant that they should not be addressed, in fact the main purpose of the resource consent application is to ensure that they are.

- 2.5 Shellfish culture has resulted in damage to ecosystems, due to over-stocking and over-exploitation (e.g. Marennes-Oleron Bay, France – Heral *et al.*, 1988; Bacher, 1989; Ria de Arosa, Spain – Tenore *et al.*, 1982 (cited Dame and Prins, 1998)). Extensive culture of shellfish (mainly mussels and oysters) can remove substantial quantities of phytoplankton, particularly when there is a high density of culture units over a large area (Barg, 1992), which is the basis of the food-web (SM3). For example, in Japan the culture of oysters has been shown to remove 76-95% of the seston (predominantly phytoplankton). In the Spanish Rias, studies of depletion rates as water passes through mussel rafts demonstrate phytoplankton is depleted as much as 35-40% (Perez Comacho *et al.*, 1991 – cited Dame and Prins, 1998).

In the Marlborough Sounds, using direct measurements at current speeds of 2-10 cm.s⁻¹ (similar to West Beatrix Bay), mussels have been found to consume 15-60% of available food and this depletion zone was shown to extend some distance downstream before recovery (Waite, 1989). Similar levels of phytoplankton depletion (up to >70% in some months) have been also found when chlorophyll *a* inside and outside existing farms in Beatrix Bay have been measured (Ogilvie *et al.*, 2000). However, the empirical estimates of Ross and Hayden (2000) for maximum phytoplankton depletion due to a 42.5 ha farm in West Beatrix Bay are given as only 23% (this is expanded to 10 to 42% by James (2000)), and is predicted to recover within a short distance of the farm. There are at least 4 factors related to these estimates of phytoplankton depletion that suggest they are underestimated and inappropriate to use for an assessment of sustainability including the

direction of water flow (discussed by Black), the variability of water flow (discussed by Black), the growth time of different phytoplankton species and the ability of chlorophyll *a* concentrations to be related to impacts on the wider environment (discussed below).

- 2.6 In terms of habitat modification, studies have been focussed on the impacts of the solid wastes, predominantly organic carbon and nitrogen, that settle to the seabed in the immediate vicinity of marine farms (e.g. Mirto *et al.*, 2000; Berelson *et al.* 1998; Barranguet 1997; Hatcher *et al.* 1994; Kaspar *et al.*, 1985; Pearson and Rosenberg, 1978). Organic enrichment of the benthic ecosystem may result in increased oxygen consumption by the sediment and formation of anoxic sediments (Smaal, 1991 – cited Barg, 1992), and the possibility of enhanced denitification (Kaspar *et al.*, 1985) and reduction in macrofaunal biomass (Mirto *et al.*, 2000), changes in abundance and species composition (Pearson and Rosenberg 1978); in extreme cases, out-gassing of carbon dioxide, methane and hydrogen sulphide (Barg, 1992) (SM2). Sedimentation under marine farms has also been shown to be much greater and therefore have the potential to cause larger negative environmental effects under farms where the water flow is weak (NCC, 1989 – cited Barg, 1992). These localised impacts can then have impacts on, and cause changes to the wider environment, which have been presented in detail by previous witnesses (e.g. changes in phytoplankton species composition due to grazing/growth rate and nutrient preferences), by transfer up the food web (SM3).

For the resource application for West Beatrix Bay, benthic impacts have been restricted to changes in the biological community and the accumulation of sediments. However, the implications and links of these impacts on the food web are not considered. In addition, the impacts under marine farms are considered to possibly be an enhancement by providing more complex habitat and food sources for a greater diversity of benthic organisms. However, no investigations of the impacts that have occurred under existing farms in West Beatrix

Bay have been undertaken. In my opinion, these farms provide an easily accessible opportunity to assess the likely impacts of a new farm in the area. My own observations under 3 mussel farms in West Beatrix Bay found little evidence of increased biodiversity of benthic organisms. Mussel shells and occasional clumps of living mussels covered by a thick layer of pseudo-faeces were the dominant feature, with visual evidence of sticky black sediments under the shells. Predatory starfish were restricted to areas that were not heavily covered by shell litter and very few, sometimes no, signs of infauna were observed in areas where there were occasional patches free of shell. These observations are consistent with severely enriched sediments, and could be expected to occur as a chronic impact of the 42.5 ha marine farm proposed for West Beatrix Bay.

2..7 In a low flow situation like Beatrix Bay (Sutton and Hadfield, 1997), which is nutrient limited and where sediment nutrient recycling has a strong influence on the dynamics of the ecosystem (Ross *et al.*, 1999), the modification of 42.5 ha of the seafloor in the northwestern head of the bay, which is the lowest flow area of the bay and is likely to be partially separated from the rest of the bay in terms of water exchange due to the anti-clockwise eddy present in this area (James, 2000; Sutton and Hadfield, 1997), it is probable that negative impacts on the existing mussel farms and the wider environment are likely to occur – as has been demonstrated by the West Beatrix Bay carrying capacity model, even with its lack of calibration and validation, limitations and assumptions (discussed below).

2.8 Recent international studies have uncovered some of the potential effects of extensive shellfish culture that have not previously been considered. For example, in the past mussels have generally been viewed as specialised herbivores. However, studies dating back to 1933 (Nelson – cited Davenport *et al.*, 2000), have appreciated that bivalves ingest zooplankton. More recently a series of studies headed by J. Davenport at University College Cork in Ireland have

demonstrated that blue mussels ingest zooplankton up to 6 mm long, as well as bind them in mucus and expel them as pseudo-faeces (Davenport *et al.*, 2000; Lehane and Davenport, 2002). These zooplankton species have been identified as mollusc eggs and larvae, polychaete larvae and small adults, bryozoan larvae, crustaceans (copepods, crabs, barnacles, amphipods, ostracods), echinoderm larvae and fish eggs. These findings have direct implications for mussel culture that will have direct impacts on local recruitment of benthic animals and pelagic fish, as well as increasing competition for primary production resources in the areas they are located (Davenport *et al.*, 2000), since zooplankton are a major link to higher trophic levels (SM3).

- 2.9 Little attention has been paid to the effect of mussels on zooplankton populations in Beatrix Bay, other than through competition for phytoplankton resources (Ross and James 1996). Davenport's work in Ireland has prompted concerns over the volumes and types of zooplankton that cultured mussels in the Marlborough Sounds could consume. For example, a study undertaken by Alex Ross in May this year showed that mussels had zooplankton in their guts, as well as fish eggs. Ross commented that it would be quite some time before NIWA could assess the ecological significance of the find, and that it was an ongoing process (Wilson, 2002). This is definitely an important factor in terms of ecosystem impacts that have so far not been considered since, in addition to the diverse planktonic algal species and zooplankton, many marine organisms spend a part of their early life history as planktonic larvae (e.g. crabs, rock lobster, bivalve molluscs, fishes, jelly fishes, echinoids, polychaetes, etc.). Indeed, the zooplankton consumption estimates of Davenport and those made from seston trawls undertaken in the Marlborough Sounds indicate very large numbers of zooplankton can be consumed by cultured mussels which will have direct impacts on local recruitment of benthic animals and pelagic fish (nearly all New Zealand coastal fish species have

pelagic eggs (Cole, 2002)), as well as increasing competition for primary production resources in Beatrix Bay.

2.10 Possible evidence of increasing competition for primary production resources has previously been presented by Ken Grange, who is the regional manager of NIWA in Nelson (Grange 1997). Grange carried out surveys of natural cockle populations in Croisilles Harbour and Delaware Inlet when NIWA was approached to provide advice on whether granting additional marine farm consent in Oyster Bay (Croisilles) may result in exceeding the carrying capacity in the bay and thus adversely affect natural marine communities, especially filter feeders that may compete with farmed mussels and oysters in the bay (Grange, 1997). The reasons for this concern centred around largely anecdotal evidence that farmed mussels in the Bay were taking longer to reach peak condition in recent years than previously, and that the cockle populations at the head of the bay had declined in recent years. Grange found differences in the population structure of the Oyster Bay cockles (mussel farms) in comparison to the Delaware Inlet cockles (no mussel farms) that prompted him to conclude that the study provided data which may give the very first glimpse that native species are being compromised and that the potential for aquaculture within the existing licenses may already be sufficient to exceed carrying capacity, to the detriment of both the aquaculture industry and native species (Grange, 1997). In Grange's opinion, the granting of additional consents may not be wise until further research had been undertaken.

2.11 The evidence presented by others in relation to changes in the shoreline species at Beatrix Bay shows close similarities with the Oyster Bay case, i.e. mussels taking longer to grow and reach peak condition and decline in natural shellfish populations. While there are several deficiencies to the Grange study, since it is a snapshot in time with no links to nutrient and phytoplankton levels, no previous studies of natural shellfish populations, etc., Grange's conclusions are based

on the relative abundances and population structures in areas where there are marine farms and where there are none.

- 2.12 Prompted by Grange's work, we recently undertook a similar study in Laverique Bay that is located on the south eastern side of Beatrix Bay (SM1). Similar to Oyster bay, Laverique Bay once provided high densities of cockles and pipis, and the anecdotal evidence suggests that the numbers of these shellfish (along with feral green mussels, seaweed and kina) have declined over the past decade. The results of 27 x 0.1 m² quadrats sampled down transects between the high and low tide range on 6 October this year, show a bimodal population structure, with a dearth of medium sized cockles indicating high mortality in this age class, very similar to that found at Oyster Bay by Grange (SM4). As can be seen from this series of graphs, the Delaware Bay population in an area absent of extensive mussel farms has a 'normal' bell-shaped population structure, while both Oyster Bay (Croisilles) and Laverique Bay (Beatrix) have few individuals in the mid-age classes, suggesting high mortality is occurring (Grange, 1997). The presence of individuals in the lower size classes indicates that settlement of cockles is still occurring (Grange, 1997).

Unlike Oyster Bay, the long-term chlorophyll *a* and nutrient monitoring in Beatrix Bay show that there has not been significant decreases in past years (not with the duration that could impact on population structure in this way since the large individuals are likely to be 10-15 years of age (Grange, 1997)), which was put forward by Grange as a way of conclusively showing whether this was the reason for decline. It is difficult to ascertain whether this is a local effect (e.g. due to nearby farms), a bay wide condition, or a state of a much larger area that is linked to climatic variables. Even so, the same conclusions and advice advocated by Grange are warranted, i.e. it is likely that native species are being compromised to the detriment of both the aquaculture industry and native species. A precautionary approach should be taken and the granting of additional consents may not be wise until

further research into the impacts of mussel farms on the wider Beatrix Bay ecosystem have been undertaken.

- 2.13 As the biomass of bivalves increases, as it is in Beatrix Bay through the continual addition of mussel farms, the matter and energy necessary to maintain these animals increases proportionally at a greater rate (Dame and Prins, 1998). This will have direct effects on the existing ecosystem e.g. extraction of phytoplankton and zooplankton (feeding and pseudo-faeces), extraction/loss/changes of nutrients (harvesting and changes to nutrient cycling and dominant nitrogen source), habitat loss and modification (under farms), etc. In Beatrix Bay these effects may be bay-wide (e.g. nutrient limitations), although magnified further into the bay due to the low current (i.e. poor circulation and flushing, re-circulating eddy in the north western corner) and under mussel farms (e.g. loss/change of nutrient cycles and habitats), or limited to species with particular feeding behavior competing for phytoplankton (e.g. zooplankton, tubeworms, bivalves, brachipods, some crustaceans), mid-water feeders competing for zooplankton (kahawai, mullet, wrasse) or bottom feeders (skate, rays, pig fish, stargazer, witch flounder, mullet) and infauna (e.g. heart urchins, brittle stars, bivalves, worms) competing for space. These effects may then impact further up the food web to higher order predators such as kingfish, john dory, witch flounder, birds and cetaceans (SM3).

3.0 The Beatrix Bay Carrying Capacity Model

- 3.1 It is my opinion, that after having reviewed the great deal of literature (journal papers, popular articles, technical reports and expert evidence) that has been produced with respect to the Beatrix Bay carrying capacity models and the Kuku Mara proposal, that there is a common view that northern Beatrix Bay West is not a good location to put in a 42.4 ha marine farm. An interesting aspect of this case is that by his own definition of carrying capacity, James (2000) presents model

results that show that the carrying capacity of West Beatrix Bay has already been exceeded, but then goes on to state that,

“The present stocking level in Beatrix Bay has been estimated to be around 2,500 tonnes on each side and our best estimates at this stage are that the present stocking level could be doubled before yield was significantly affected”

3.2 This is contradictory to the modelling results.

3.3 Sustainable carrying capacity is the term that has been used as the measure of the Beatrix Bay ecosystem’s ability to maintain healthy organisms – once it is surpassed cultured and native species will be negatively affected. The definition that has been used through out the PGSF sustainability of shellfish fisheries programme is,

“sustainable carrying capacity is the stocking density that maximises production without negatively affecting condition and growth” (e.g. James, 2000, Hayden *et al.*, 2000).

3.4 By it’s own definition this is clearly a measure that targets cultured shellfish and does not take into account other parts of the ecosystem. SM5 shows the results of the Beatrix Bay Model that were presented by James (2000) and also by Hayden *et al.* (2000). It is obvious that in west Beatrix Bay the growth rate decreases even at low stocking levels (top graph) and that harvest tonnage begins to decrease beyond the estimated current stocking level of 2500 tonnes (bottom graph) – these graphs show that growth rate is being negatively affected and production is not being maximised, i.e. carrying capacity is already being exceeded. While Hayden *et al.* (2000) suggest that the potential for increased meat yields will need to be weighed against the longer time the shellfish must be held on the farm to achieve these yields

(decreased growth rate is contradictory to the carrying capacity definition) and that increasing the yield also needs to be viewed in the context of other potential effects on the long-term nutrient cycle and benthic communities, no such considerations have been suggested for the resource consent application for West Beatrix Bay.

3.5 The Beatrix Bay models do not adequately consider the wider environment, which is due to them being directed towards the carrying capacity of cultured shellfish. The models are structured as tools to enable the maximisation of cultured shellfish production within the bounds of the environmental parameters that have direct impacts on the culture of shellfish (nutrient inputs, water flow, competition with zooplankton, mussel energetics, etc.). This becomes clearer when the food-web diagram is compared to one that defines the parameters that the Beatrix Bay models address (SM6) – many parts of, and interactions with, the wider environment are not represented.

3.6 Ross and Image (2001) state, “the sustainable use of the marine environment has two facets in an ecological context:

- 1) the sustainable use of the resource so as not to cause significant effect on the production of existing shellfish farms, often termed sustainable production or carrying capacity; and,
- 2) perhaps more significantly, not to have an adverse environmental effect such as an undue influence on the functioning or structure of the marine ecosystem.”

3.7 While environmental and production sustainability are closely linked, the model and definition do not go far enough to consider the wider environment and minimisation of environmental effects. James argues that while the definition of carrying capacity relates to shellfish, in the current absence of very detailed and comprehensive ecological studies, it is probably the best indicator of healthy water column ecosystem and since other components of the biota rely on phytoplankton, then this is an indicator of the state of the whole environment. If we accept this assumption, then we have to accept

that the West Beatrix Bay ecosystem has already surpassed its sustainable carrying capacity, since according to the model results the growth and productivity have already decreased. Anecdotal evidence of mussel farms and the bay, and the scant data that is available on native shellfish in Beatrix Bay all support this. In addition, the methods used to indicate a healthy water column (i.e. chlorophyll *a* concentration) are not adequate for this purpose (discussed below).

3.8 It is also important to note that the carrying capacity models divide Beatrix Bay east-west. This means that the carrying capacity estimates are based on the whole side of the bay. This is misleading, especially with respect to the hydrodynamics of the bay, because the mouth of the bay can be attributed with a much larger proportion of the total estimated carrying capacity than the northern head of the bay due to the higher flow rates and its proximity to 'new' water entering the bay. As has previously been pointed out by Black, at a minimum the bay should be divided into 4 sections, and if it is divided into only two, then the division should be north-south. Qualitatively, observations of the native marine biota around the intertidal and subtidal margins of the bay show a clear north-south trend, with relatively lower species diversity and abundances of marine organisms in the north of the bay compared to the south (Mead, 2002).

3.9 The inadequacies of the model in addressing the wider ecosystem are again highlighted by Grange's (1999) definition of sustainability. Grange (1999) identified 3 aspects of sustainability important to the mussel farming industry.

1. Farm sustainability – how many mussels will the farm grow, how fast, and what will be the quality?
2. Fishery sustainability – will the establishment of large numbers of mussel farms affect the carrying capacity of nearby farms?
3. Ecological sustainability – will the establishment of large numbers of mussel farms affect the ecological processes in the surrounding environment?

The first aspect is addressed by individual farm practice (e.g. stocking densities) and locality, the second is addressed by the Ministry of Fisheries during the licence application process, and the third by the Resource Management Act during the consents process, although these are not mutually exclusive (Grange, 1999). In my opinion, the impact assessment that supports the application for resource consent for a 42.5 ha marine farm in West Beatrix Bay does not adequately address aspect 3 as required for it to be granted. By its own definition the carrying capacity model is designed to consider only 1 and 2 (sustainable carrying capacity is the stocking density that maximises production without negatively affecting condition and growth), and not the ecological processes of the wider environment. The recently released Ministry of Fisheries draft “A Guide to Preparing a Fisheries Resource Impact Assessment” notes the deficiencies of shellfish carrying capacity models in that they do not consider minimising environmental effects (Ministry of Fisheries, 2002).

- 3.10 The focus on mussel production, the lack of full consideration of wider ecosystem links and the gross simplification of environment (e.g. treating the whole of western Beatrix Bay as one unit) all suggest that in terms of sustainability of the wider Beatrix Bay ecosystem, the stocking estimate given by James (an additional 2,500 tonne on the west side of Beatrix) is an overestimate.

4.0 Monitoring and Adaptive Management

- 4.1 In my opinion there are aspects of the methods proposed to monitor to assess the environmental impacts of the proposed West Beatrix Bay marine farm that are flawed. The conditions of the resource consent present chlorophyll a measurements and benthic surveys as monitoring measure to evaluate the farms impact on the environment and other marine farms. Measuring chlorophyll a concentration is an unreliable way to measure water column impacts, especially at the generally low levels present for the majority of the time in Beatrix Bay.

- 4.2 Measurements of chlorophyll *a* pigment can be used as an estimate of phytoplankton biomass, which is the most important food for mussels and other filter feeders (James, 2000). However, while chlorophyll *a* levels show good correlation to factors such as nutrient inputs, temperature gradients and river flows (Ross *et al.*, 1998b), as well as phytoplankton depletion (Ogilvie *et al.*, 2000), there is little evidence of a good correlation between mussel growth and condition in Beatrix Bay and chlorophyll *a* concentrations. Indeed, the published data on chlorophyll *a* concentrations and average harvesting yields shows that yields are often highest when chlorophyll *a* levels are lowest (Zeldis and Pinkerton, 2000). In fact, chlorophyll *a* is not well correlated to mussel growth. Moreover, chlorophyll *a* concentration is not the best method of measuring the abundance of phytoplankton (Ross *et al.*, 1998b), and it does nothing to address the mix of species present, which have been shown to change significantly through the seasons in the Marlborough Sounds (Zeldis and Pinkerton, 2000; Gall *et al.*, 2000). This is an important issue, since chlorophyll *a* levels are used as an indicator of phytoplankton depletion, which are then related to impacts on other farms and the wider bay ecosystem (James, 2000).
- 4.3 Measuring phytoplankton abundance by the concentration of chlorophyll *a* is undertaken for its simplicity in comparison to tedious cell counts (Ross *et al.*, 1998b). However, it is well known that different species of phytoplankton have different nutritional value (Grant and Bacher, 1998), and some may even inhibit shellfish filtration (Prins *et al.*, 1994). It is food quality that is more important than quantity in accurately predicting growth, and food quality is dependent on species (Grant and Bacher, 1998; Campbell and Newell 1998). Even so, chlorophyll *a* concentrations are being used as a major constituent of a carrying capacity model and as the main parameter monitored to address impacts of farms. A better and more appropriate measure is needed, such as the cell counts and conversions to carbon suggested by Ross *et al.*, (1998) for long-term studies. Using chlorophyll *a*

concentrations to manage and monitor the impacts of mussel farms has been likened to monitoring the weekly family supermarket bill as an indication of how much weight the people in the family will gain – the total bill gives no indication of the nutritional value of the food that has been bought. Monitoring should include control sites (as suggested by Ross and Image, 2002), but also consider monitoring the growth and condition of the shellfish (not just phytoplankton abundance), since healthy mussels have consistently been related to the health of other biota in the bay ecosystem.

- 4.4 The composition of the water column is undoubtedly modified (changes to the nutrient composition, phytoplankton composition, zooplankton composition, etc.) when water passes through a mussel farm. However, for the 42.5 ha marine farm in West Beatrix Bay only depletion rates are estimated and, based on mixing with undepleted water and phytoplankton growth, these are said to recover within a few hundred meters of the farm boundary (James, 2000; Ross and Hayden, 2000). However, this is based only on measurements of chlorophyll *a*, when it is well known that changes in soluble nutrients (e.g. increased ammonium) occur prompting changes in phytoplankton species composition (Gibbs and Vant, 1997) as those that have been consumed by the mussels are replaced (this is detailed in the evidence of Longmore). This is further complicated by differences in growth time for the range of phytoplankton species that inhabit the Beatrix Bay waters (hours to days). Gibbs (2002) stated that,

“... phytoplankton most likely to take advantage of these released nutrients may not be the same species as those extracted by the mussels in the first place. Furthermore, opportunistic phytoplankton in this case are likely to be smaller, fast-growing species that have less nutritional value than those selectively extracted by the mussels.”

- 4.5 Both of these factors lead to changes in phytoplankton species composition, which affect the nutritional value of the total phytoplankton (or chlorophyll *a* concentration). It is not appropriate to base the majority of impact assessments and farm management on chlorophyll *a* measurement – the composition of the phytoplankton is likely to be far more important, which has been shown to be reliant on nutrient species, stratification, water temperature, etc. (Ross *et al.*, 1998). If the species composition was changed in a way that was beneficial, then the downstream mussel farms and natural biota would be thriving. However, since the reverse has been observed to be true, it is reasonable to suggest that the changes are not conducive with enhanced food supplies. In my opinion there are some important issues that need to be addressed with respect to the evidence that suggests that phytoplankton is restored quickly once the water has passed through a farm based only on chlorophyll *a* concentration, since this measure does not account for the changes to the phytoplankton species composition which has been shown to be the most important measure of water column impact.
- 4.6 Chlorophyll *a* is low in the Beatrix/Crail/Clova Bay complex in comparison to other parts of the Marlborough Sounds, which is related to nutrient levels (Gibbs *et al.*, 1992). In fact, during the spring/summer when dissolved inorganic nitrogen levels are at their lowest, Ogilvie *et al.* (2000) suggest artificially increasing nitrogen concentrations within the mussel farms in Beatrix Bay to increase growth rates. It has been reasoned that the consistency of chlorophyll *a* levels inside and outside of mussel farms (once they have mixed and new growth has occurred) is evidence that there is no impact on the productivity of the water column. On closer examination of the work that has been done to support this (e.g. Ross and Image, 2002), it is clear that the ambient chlorophyll *a* levels are already low (0.5 – 1.2 ug/l) and the flow regime is unstable (i.e. not uni-directional through a farm). Indeed on the 2nd September this year, the first presentation at the New Zealand Marine Science Society conference in Nelson by Mark Gibbs of the Cawthron

Institute showed results of a similar study as that carried out by Ross and Image (2002). In this study chlorophyll *a* levels inside a farm were shown to be less than half of those outside the farm 5.5 v 2.5, and similar depletion was shown to occur due to the cumulative effects of water flowing through several farms (uni-directional) and to persist for over 500 m beyond the farms.

- 4.7 There are still many uncertainties with respect to the use of chlorophyll *a* concentration for monitoring and assessment and the issues of recovery from phytoplankton depletion. At present this may be the best measure available, but I believe in the present case, where there is evidence that suggests a sustainability problem, it is not sufficient. Following the logic of James (2000), since there is no detectable reduction in phytoplankton 80 m from a 3 ha marine farm, covering the whole of Beatrix Bay with 3 ha mussel farms spaced say 200 m apart would give sufficient time and space for mixing and regeneration to ambient levels. However, the models show reductions in 'carrying capacity' at the current levels?
- 4.8 Better linkages between these issues and mussel growth and condition may be found looking more closely at the species composition – both phytoplankton and zooplankton species composition has been shown to have strong seasonal cycles (Zeldis and Pinkerton, 2000). In addition, the cumulative effects associated with phytoplankton depletion and changes to species composition due to multiple mussel farms in enclosed embayments such as Beatrix Bay, needs to be better addressed. This is in itself very difficult, since as pointed out by Ross and Image (2002), it is clear that at present from the information available it is not possible to establish with certainty the likelihood of cumulative effects. Analysis of mussel industry productivity data may be able ascertain whether the trend shown by Mr. King, of decreased carrying capacity in the head of Beatrix Bay compared to little change outside the bay in 'better' water, is a phenomenon that is occurring all over the Sounds. It is logical to assume that if the threshold of carrying

capacity in the Sounds is being exceeded that the least productive, relatively isolated areas of low water flow and flushing such as the heads of bays are likely to be the first effected.

- 4.9 The monitoring proposed is linked to the concept of adaptive management that is a concept that should be viewed with caution, especially in areas that show signs of environmental stress. Changes to the ecosystem, such as those that can occur through the modification of the environment caused by a large marine farm may not be reversible. For instance, conditions may become more favourable for a different species than previously occupied a certain niche and so a species change occurs. Once the new species becomes established it may then be able to persist and out-compete the original component of the ecosystem, even if the original modifying factor (e.g. a marine farm) is removed. Such changes are extremely hard to foresee. For example, Chesapeake Bay in the USA is a large estuarine water body with an estimated residence time of 22 days (similar to Beatrix Bay). Removal of oysters from the bay due to harvesting, disease and declining water quality resulted in numerous bivalves invading the bay, into areas once dominated by oysters. Studies have shown that the bivalves in the bay now remove up to 50% of the annual phytoplankton primary production in the upper bay. It is now believed that restoration of oysters into Chesapeake Bay to reverse the change in water quality will not be successful because they have already been functionally replaced by other bivalve species and due to anaerobic conditions in the deeper portions of the bay (Gerritsen *et al.*, 1994 – cited Dame and Prins, 1998). Similar, unforeseeable, ecosystem changes are possible in the Marlborough Sounds and may already be occurring.

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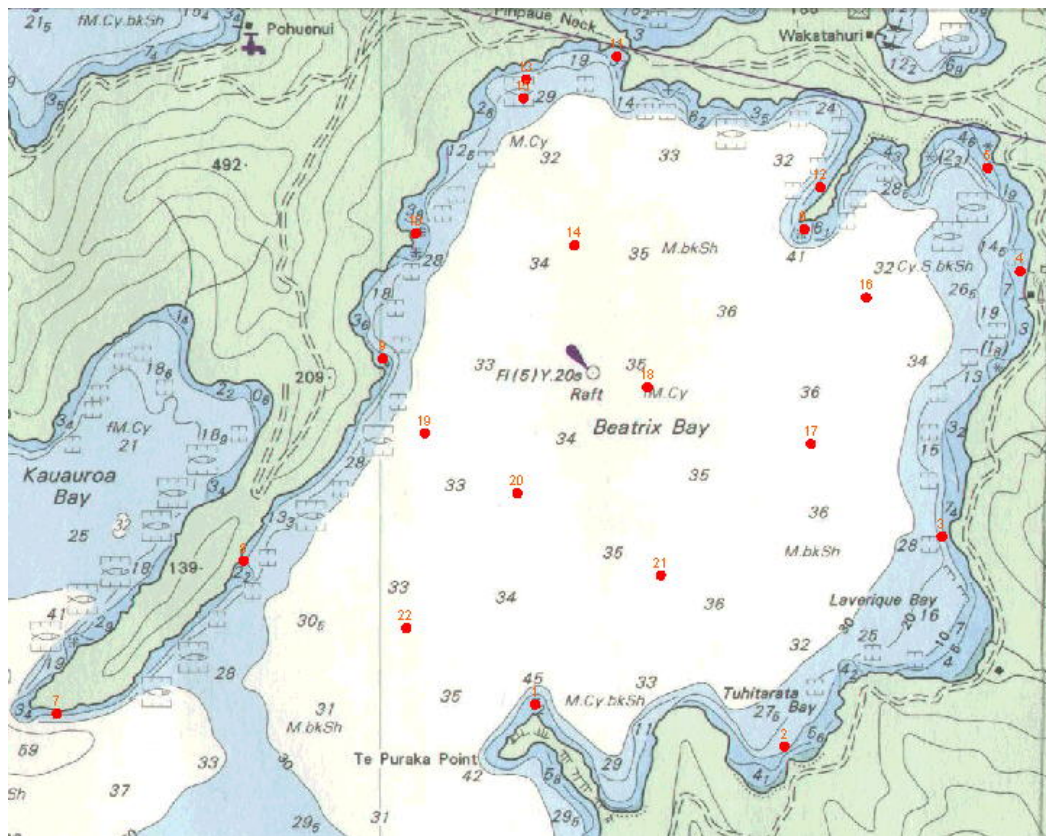
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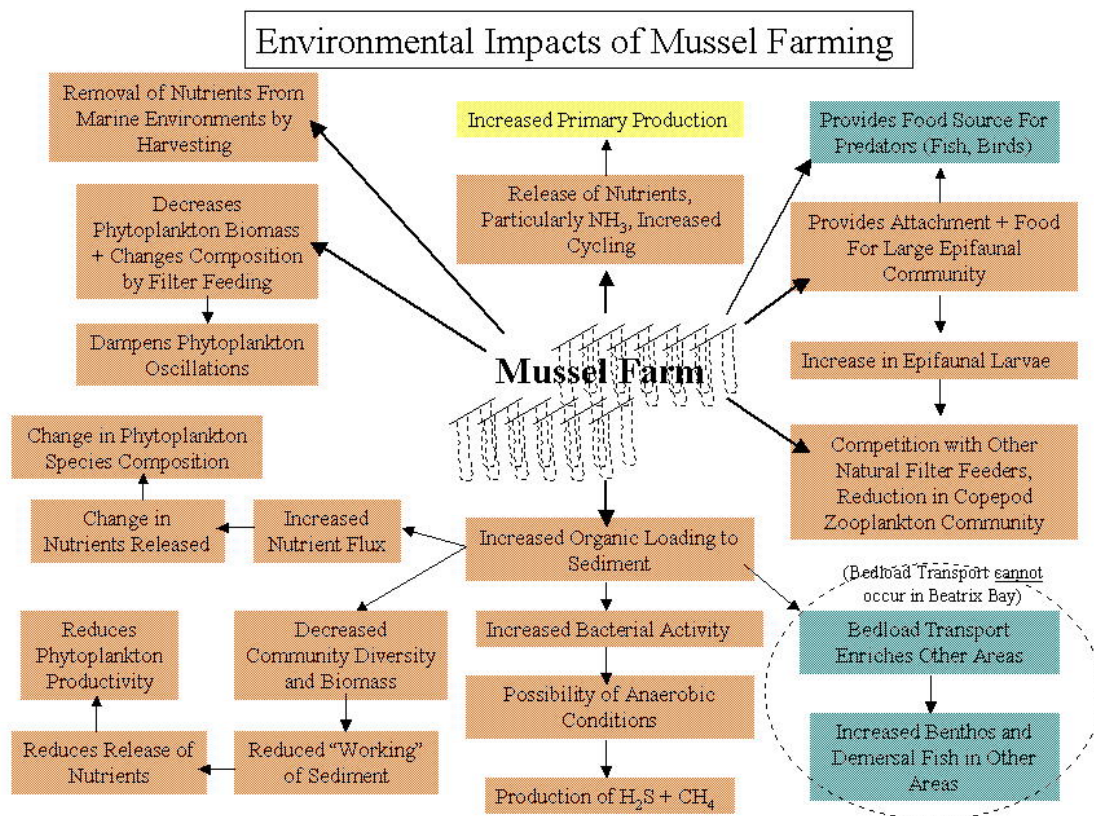
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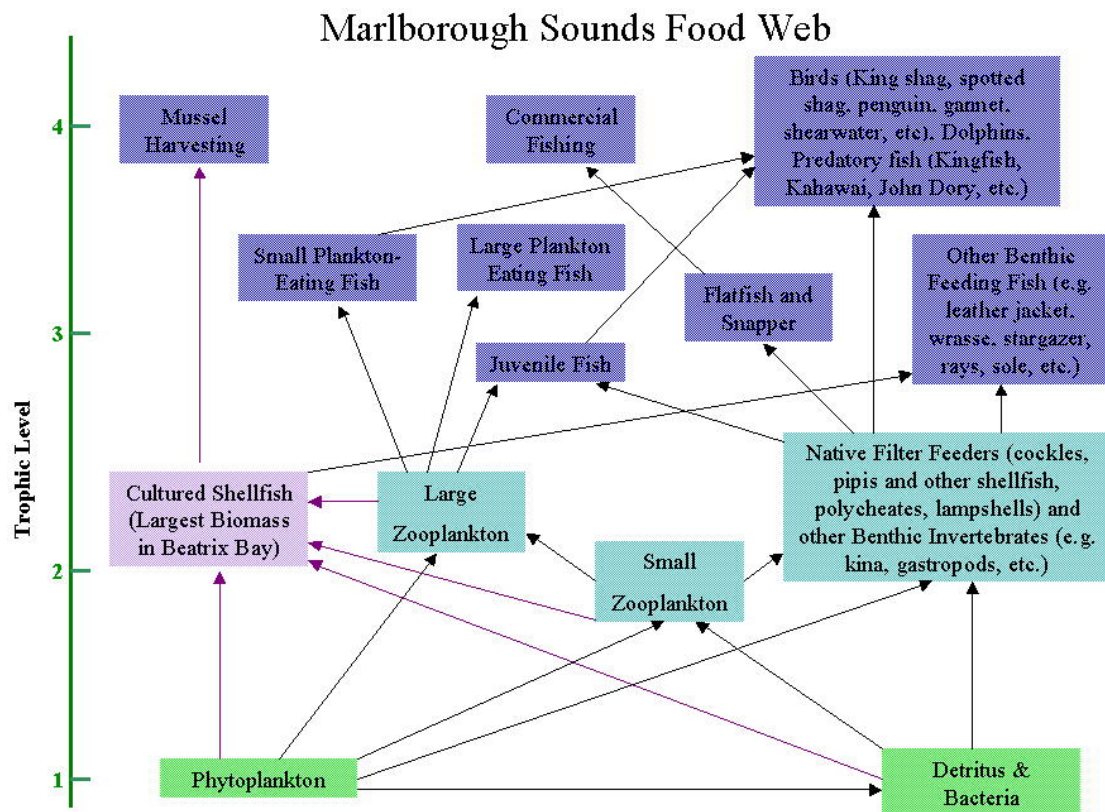
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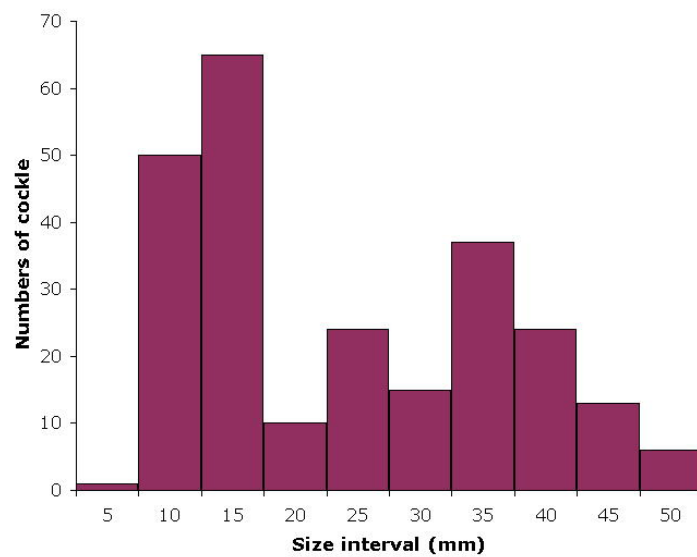
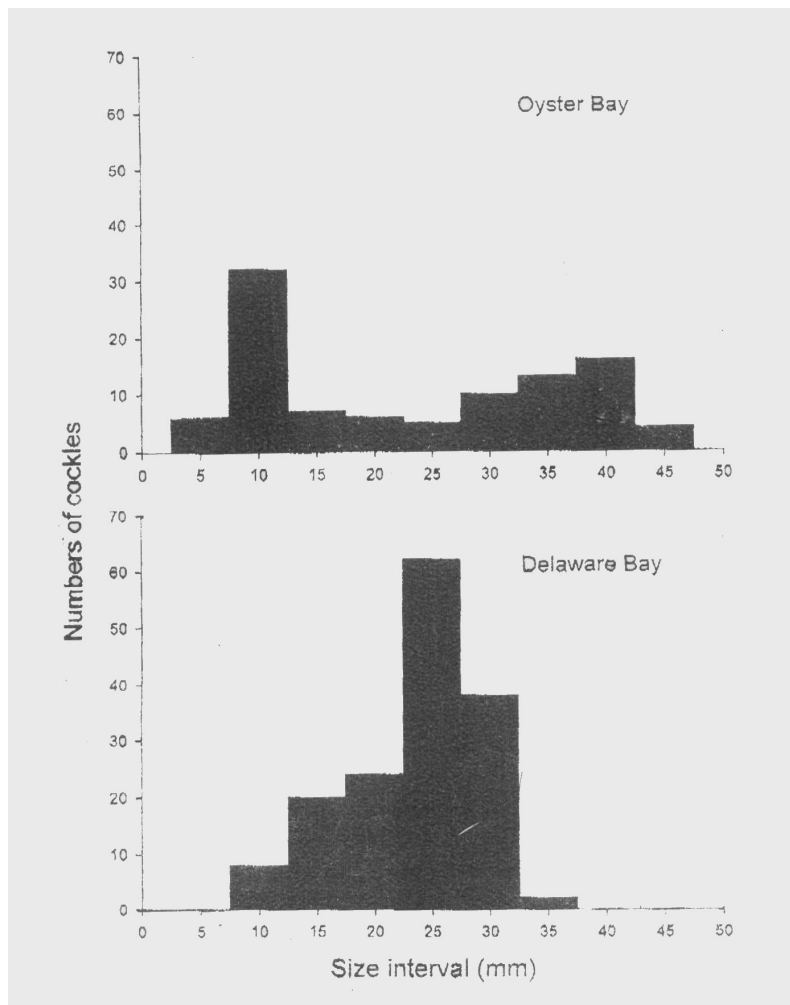
SM1 - Survey sites



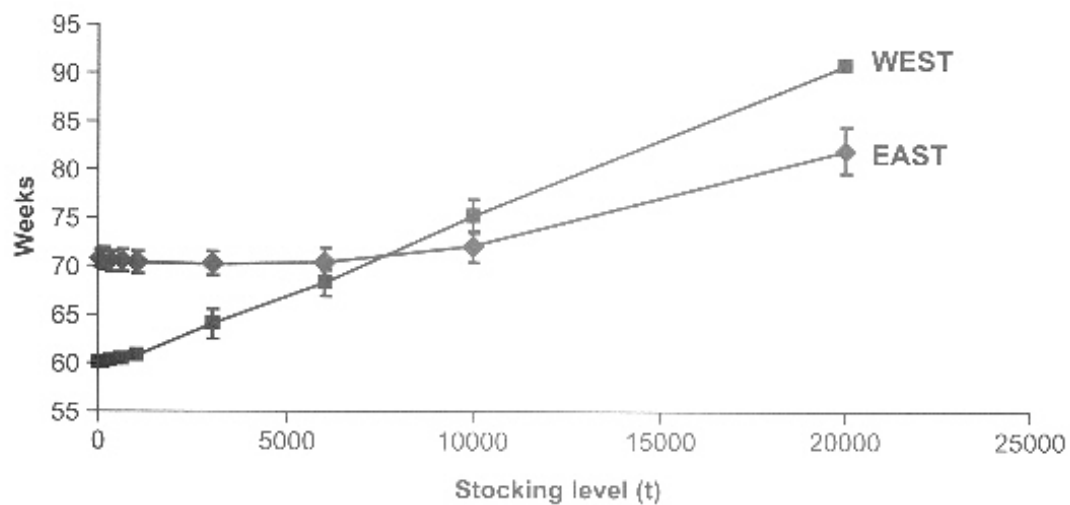
SM2 – Summary of the possible ecological effects of mussel farming. Note, some effects are contradictory, and not all effects will be seen at one site (adapted from Barg, 1992).



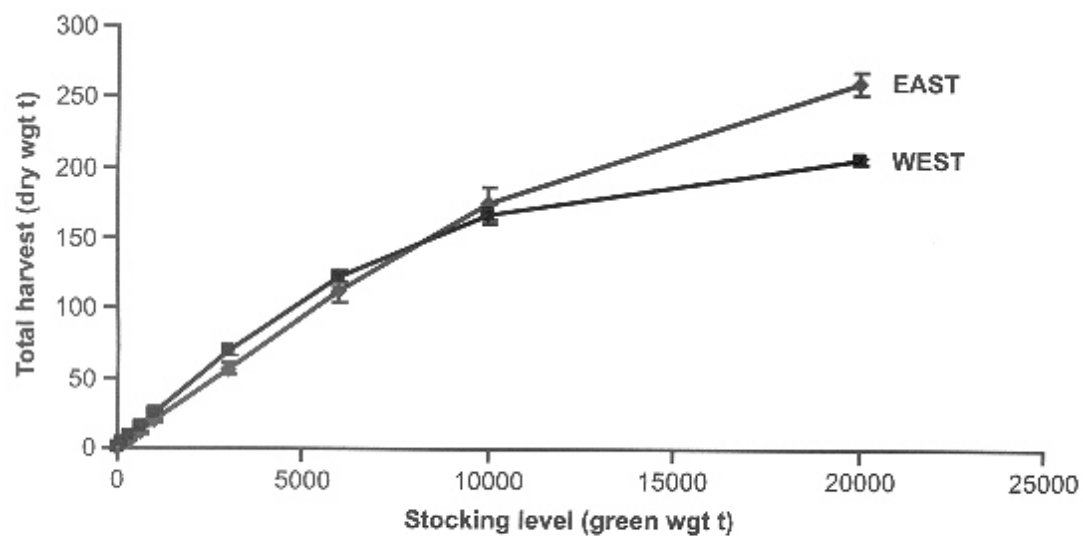
SM3 – Marlborough Sounds marine food web (adapted from Bradford-Grieve, 2002). Note the range of food linkages between cultured mussels and the lower trophic levels.



SM4 - Population structure of cockles at Oyster Bay (top), Delaware Bay (Middle) and Laverique Bay (bottom).

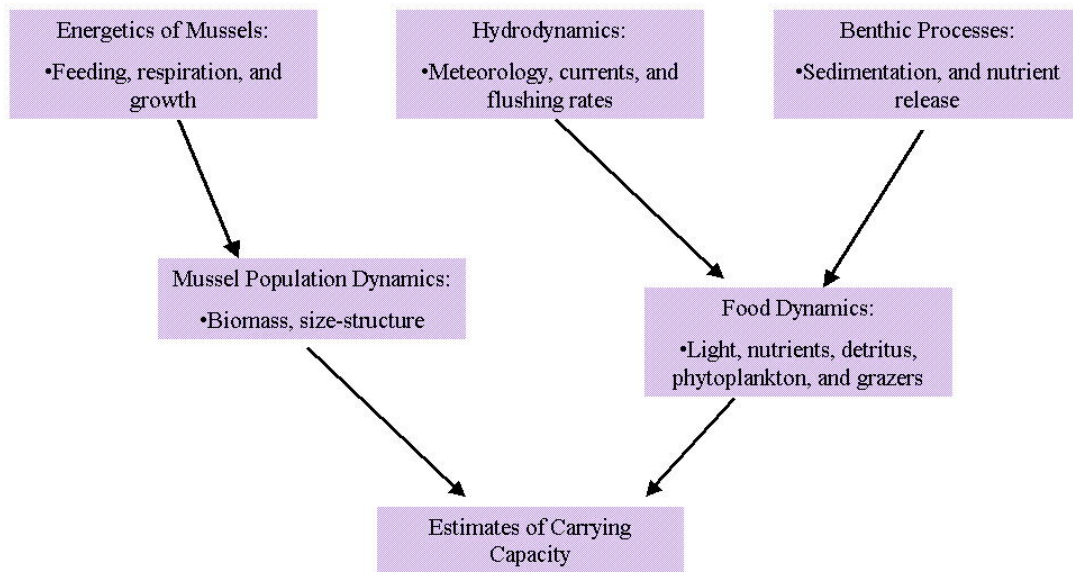


Predicted time for mussels to reach harvest size in west and east Beatrix Bay at different stocking levels



Predicted harvest from west and east Beatrix Bay at different stocking levels

SM5 – Results from the Beatrix Bay carrying capacity model



SM6 – Schematic diagram of the submodels and parameters used to develop the carrying capacity model for Beatrix Bay (Source, James and Ross, 1996).

Attachment 4

STATEMENT OF EVIDENCE

Andrew Robert Longmore 10 October 2002

STATEMENT OF EVIDENCE

Statement of evidence of **Andrew Robert Longmore**. 10 October 2002

1. My name is Andrew Robert Longmore. I live in Queenscliff, Victoria, Australia. I hold the degrees of Bachelor of Applied Science and Master of Applied Science from the University of Melbourne. I am a marine chemist, employed by the Marine and Freshwater Resources Institute, a research arm of Fisheries Victoria, since 1980. While my evidence is based on my experience, the views I express are my own, and not necessarily those of my employer.
2. My area of expertise involves water quality studies to improve our understanding of natural and human impacts on nutrient cycling in estuaries and coastal water. This has involved the development and application of nutrient mapping systems to identify impact zones around coastal discharges, and innovative techniques to study the recycling of nutrients from sediments. These techniques have significantly increased our understanding of nutrient cycling in Port Phillip Bay and the Gippsland Lakes (Victoria), Moreton Bay (Queensland), Swan River and Wilson Inlet (Western Australia) and the Derwent Estuary (Tasmania), and provided the backbone on which predictive models have been built. They have most recently been used to study nutrient cycling in Australian prawn farms. My research has been published in about 20 peer-reviewed scientific journals and numerous technical reports. I have been used as an external reviewer for the NZ Journal of Marine and Freshwater Research.
3. In reaching its decision that the impact of the proposed new farm is “likely to be no more than minor”, the resource consent hearing

accepted NIWA (Dr Mark James) assertions that: (a) Carrying capacity is greater than the sum of current and proposed new production; (b) Benthic impact of a new farm is likely to be small; (c) Staging will avoid an impact on other farms; and (d) Mussels are good indicators of bay health. I believe the panel should not have accepted these assertions.

4. **(a) Carrying capacity.** Models are, by their nature, gross simplifications of naturally complex systems. We can have no confidence in the output (predictions) of models unless and until they can be shown, as a minimum, to reproduce field observations. NIWA has chosen to model only selected broad functional categories of the ecosystem (Ross et al. 1999). The NIWA model, from which carrying capacity for Beatrix Bay was estimated, includes modules to reproduce hydrodynamics, phytoplankton growth and nutrient dynamics, and mussel growth and condition (NIWA 2000a). Ross et al. (1999) stated that “the applicability of such models for management decisions, when not calibrated and tested, is questionable”. No evidence has yet been produced that the NIWA model accurately reproduces any of these variables. At the time of James’ submission that current stocking in Beatrix Bay is well below the carrying capacity, NIWA (2000a) admitted that they were still validating the model. Certainly no details of model calibration have been published for peer review. We believe that some model parameters used in the NIWA model are inherently inaccurate, and other important parameters have been left out completely. It is my opinion that in the absence of full calibration, the model output is not useful as evidence, or for making sound judgements about carrying capacity.

5. Dowd (1997) warned that simple ecological models of cultured bivalve growth are limited in their predictive capacity because of high sensitivity to small changes in physiological parameters describing the mussel energy budget. Grant and Bacher (1998) came to a similar conclusion, using the example of the digestibility of food. In Canada, the ratio of particulate organic carbon to particulate organic matter (POC/POM) varies naturally by 300% during the mussel growth cycle. However, in the models they developed, a 10% change in this ratio (digestibility) led to a 40-60% change in predicted mussel weight after 8 months. The situation may be even more uncertain with the NIWA model, since POC/POM varied by more than 500% over 24 hours in Kenepuru Sound (Hawkins et al. 1999).

6. Grant and Bacher (1998) found that total chlorophyll measurements do not account for temporal changes in the composition of phytoplankton communities and their differing digestibility. They concluded (and Campbell and Newell 1998 concurred) that food quality is more important than quantity in accurately predicting growth. Hawkins et al. (1999) found that retention efficiency of organic matter and chlorophyll each varied strongly with both the abundance and composition of available seston, and accurate estimation of both were critical to the development of further model parameters. As far as we can tell, the NIWA model uses total chlorophyll with a fixed digestibility, and takes no account of varying availability or organic content. Ogilvie et al. (2000) stated that the relative nutritional value of phytoplankton from different depths in Beatrix Bay is unknown, but NIWA assumes the plankton in deeper water to be accessed by

the proposed farm has the same nutritional properties as that in the surface layer.

7. Campbell and Newell (1998) believed that “an accurate determination of the carrying capacity for mussel aquaculture within the larger ecosystem would require the modelling of system components such as seaweed, zooplankton, macrobenthos, wild mussels, etc.”. Zooplankton appears to be the only one of these components included in the NIWA model. Grant et al. (1998) found that mussel fouling comprised about 25% of the nutritional demand of a mussel farm, but there is no evidence the NIWA model takes such a large demand into account. Indeed, the experiments of mussel growth on which the NIWA model is based (Hawkins et al 1999) were carried out with mussels which had been cleaned of all epibiotic growth. Odum et al. (1983), in one of the earliest simulations of mussel culture in Marlborough Sounds, predicted that stocking at a level to maximise profits would lead to a reduction in wild mussels, which are the source of the spat necessary for mussel culture. There is already evidence of poor and variable spat fall at current stocking levels. Spatfall statistics in Beatrix Bay, to be presented by another witness, and the studies of Bayne (1976) show spatfall reductions, possibly due to stressed mussels during prolonged periods resulting in an increase in abnormal embryonic development.

8. Variables modelled by Hawkins et al. (1998) as inputs to the NIWA mussel nutrition sub-model included total organic content, clearance rate, retention efficiency and net absorption rate. Between 34 and 47% of the variance in these variables was not

explained by the models derived. In other words, a large proportion of the natural variability of measures of mussel food supply and growth is not accounted for in model derivation. In addition, all of these relationships were derived for mussels of a single size; we cannot assume that the same relationships derived for one size apply over the whole growth cycle. For example, Waite (1989) found that the maximum growth efficiency of mussels in the 30 mm length class was 0.48 compared to 0.66 in the 80 mm length class. I believe that the modelling is in its infancy, lacking peer-reviewed publication and solid calibration. The modelling results presented by James (2000) are therefore potentially misleading.

9. James' exhibit MRJ 20 (estimated harvest weight *vs* stocking level) indicates that stocking in east Beatrix Bay could be increased to at least 10,000 t before yield declined. In contrast, some curvature is predicted for west Beatrix Bay, even at current stocking levels. This implies that stocking in the east Bay could increase significantly without affecting current production, whereas additional production in the western bay will affect current producers (by extending the time taken to grow). Nevertheless, the tribunal rejected the application for the eastern farm, because they had "sufficient information to say there would only be sufficient nutrients and food to sustain one farm". MRJ20 indicates farms on the western side are more vulnerable to impact from additional production than those on the eastern side. The NIWA estimate of carrying capacity relates only to the capacity to produce cultured mussels; it ignores completely the possibility of changes to other components of the ecosystem.

10. Hawkins et al. (1999) observed wasting of mussels at chlorophyll concentrations below $0.86 \mu\text{g L}^{-1}$, and no significant growth below $1 \mu\text{g L}^{-1}$. Given that there have already been substantial periods during 1996-98 of chlorophyll concentration below $1 \mu\text{g L}^{-1}$ in Beatrix Bay, perhaps driven by meteorological or climate change factors, we can have no confidence that the food supply will be adequate for existing farms, much less additional ones. The idea of therefore supporting increased stocking is difficult to justify.
11. **(b) Benthic impact.** James used a video showing sea stars feeding on fallen mussel clumps to base his assertion that the proposal would have a minimal impact on the benthos. However, the major impacts are likely to be somewhat less visible. They may include changes to nutrient recycling from the sediment, and changes to sediment fauna. Mussel culture leads to enhanced sedimentation of organic matter (Barranguet 1997), and the rate of sedimentation is linearly related to chlorophyll biomass (Hatcher et al. 1994). In a general sense, increased organic enrichment leads to a reduction in the number of species living in the sediment, but possibly an increase in total abundance of a few opportunistic species, usually worms (Pearson and Rosenberg 1978).
12. Mitro et al. (2000) observed a significant decline in meiofaunal abundance under a mussel farm, though the implications of this to the wider ecosystem are not clear. Kaspar et al. (1985) found sediment under a small (1.5 ha) farm was enriched in organic nitrogen and ammonium, while the benthic community was depauperate compared to a control site. Denitrification (conversion of nitrate to N_2 gas) was $\sim 20\%$ higher in sediment at the mussel farm than at the reference site, and was 10 times higher in the detritus-covered mussels at the farm than in the

reference site sediment. Enhanced denitrification leads to a net loss of nitrogen from the system. The extent of denitrification is a balance between the rate of supply of organic matter, its bacterial breakdown, and the supply of oxygen, which may be mediated by infauna irrigating the sediment (Berelson et al. 1998). The impact of increasing farm size (from 2-3 ha to 42 ha) on denitrification is unknown. If an increasing area of sediment enriched in organic matter below a much larger 42 ha farm has no impact on the infauna, the increased size may lead to increased denitrification (and loss of nitrogen from the ecosystem), which may impact on all farms in the Bay. Alternatively, the organic deposition over a wider area may cause a reduction or loss of the infauna, leading to a switch from denitrification to ammonium release. Whether this is a benefit to mussel production or not will depend on whether the ammonium is taken up by beneficial plankton, or those of low nutritional value. Bear in mind that sediment nutrients are released to the bottom waters, which are already nitrate-rich, and apparently inaccessible to surface-dwelling plankton in summer. However, if the sediment-released nutrients do stimulate a bloom of non-beneficial algae, the dominant bottom current will carry the nutrients (or bloom cells) toward the inshore farms. The large farm may well produce an impact on other farms, while escaping such impacts itself.

13. In a Canadian study, the largest response of the sediment community to suspended mussel culture was increased ammonium release from the sediment year-round, with the highest rate in summer (Hatcher et al. 1994). There was a negative relationship between bottom water nitrate concentration and ammonium flux, which Hatcher et al. (1994) took to indicate

ammonification (conversion of nitrate to ammonium) and denitrification (conversion of nitrate to N₂ gas), both processes reducing the concentration of the nitrogen species thought most critical for phytoplankton in Beatrix Bay.

14. Changes in sediment nutrient cycling in Beatrix Bay are important, because the nitrogen supplied by sediment release is of a similar magnitude to Cook Strait input, and much larger than river flow or mussel excretion inputs (Gibbs et al. 1992). Ross et al. (1999) confirmed that sediment nutrient recycling has a strong influence on the dynamics of the Beatrix Bay ecosystem.

15. NIWA (2000) admits that “increasing the yield also needs to be viewed in the context of other potential effects of the long-term nutrient cycle and benthic communities”. No such context was provided in James’ submission to the application process; they were rejected completely. Furthermore, the estimates of carrying capacity are concerned only with production of cultured mussels, and completely ignore any impact on other fauna.

16. **(c) Water column impact.** Bradford et al. (1987) indicated that poor mussel condition was a periodic problem in the early 1980s, which was attributed to food shortage due to nutrient depletion in summer. Currently about 160 ha of Beatrix Bay is farmed, with 13 ha used for spat collection; the proposed farm increases the area by 24%. Ross et al. (1998) believe that inner sound farms are more affected by natural hydrographic variation in nutrient supply than those closer to the sound mouth.

17. Ogilvie et al. (1998) observed occasions when chlorophyll concentration was higher inside farms than outside. They attributed this to phytoplankton growth enhanced by ammonium excreted by mussels. At first glance, this may be seen as an advantage, leading to higher food production, and therefore potentially higher mussel growth. However, Prins et al. (1994) noted that mussels reduce clearance rates if the available phytoplankton are not suitable food. Beatrix Bay is an ecosystem in which plankton were originally dependent on riverine and oceanic nitrate, and it is not surprising that growth of the endemic plankton, dominated by diatoms and dinoflagellates, is stimulated more by nitrate than ammonium (Gibbs and Vant 1997). However, an ecosystem in which increasing amounts of ammonium (from mussel excretion) become available may suit non-endemic phytoplankton species which are unsuitable as a food source. For example, *Phaeocystis* sp, the dinoflagellate *Gyrodinium aureolum* and the chrysophycean *Aureococcus anophagefferens* have all inhibited mussel filtration (Prins et al. 1994), possibly by clogging the gills with mucus. Rhodes et al. (1995) reported *Phaeocystis* blooms in NZ waters in 1981. Coccolithophore blooms in 1992 were associated with fish mortalities in Big Glory Bay, NZ, and growth in culture was enhanced by ammonium addition. Similarly, a raphidophyte (*Heterosigma carterae*) bloom in Big Glory Bay in 1989 was associated with increased nitrogen supply (from fish farming), and has led to fish kills internationally (Chang and Page 1995).

18. The concern is therefore one of enhanced ammonium excretion from a large-scale mussel farm leading to a shift in the phytoplankton community to less edible species.

19. **Staging.** Staging can be considered with respect to chlorophyll depletion and ammonium production.

20. **Chlorophyll depletion.** In his submission to the tribunal, James admitted that he cannot estimate depletion of food supply by such a large farm. All previous measurements have been on much smaller farms. One problem he faces is that water flow is restricted through even small farms: Gibbs et al. (1991), Boyd and Heaman (1998) and Karayucel and Karayucel (1998) observed reductions in flow through farms of up to 70% compared to currents outside the farms. Feeding efficiency (and therefore carrying capacity) will be lower under such conditions than if flow reductions are ignored. There does not appear to be any scope for the tribunal to reduce the stocking rate on the proposed farm , if the first stage is found to have a significant impact on the other farms. Ogilvie et al. (1998) found chlorophyll depletion of up to 72% in small farms. No one has been willing to estimate depletion in a farm 10-20 times larger, but James proposes simply to measure it once the farm is constructed.

21. **Ammonium production.** Mussels excrete ammonium, which may fuel increased plankton growth (Barranguet 1997). It is possible that ammonium produced by many small, widely-spaced farms is dispersed by mixing with water between the farms before relatively high concentrations confer a competitive advantage on

one species of phytoplankton over another. On the other hand, one large farm is more likely to produce a significant area of high ammonium concentration, making algal blooms more likely. No one is in a position to know whether such blooms would be of edible or inedible plankton.

22. **(d) Mussels as indicators of bay health.** James asserted that “if mussels are doing well, other biota should also be doing well”. Hawkins et al. (1999) observed that wasting occurred for mussels supplied with less than 0.86 ug L^{-1} chlorophyll, and that significant growth could only be expected for chlorophyll concentrations above 1 ug L^{-1} . Chlorophyll concentrations below 1 ug L^{-1} were experienced in Beatrix Bay for considerable periods during 1996-98, during which periods mussels did not do well. By James’ definition, the Beatrix Bay ecosystem has been unhealthy for considerable periods. While the cause(s) of the poor growth may be natural oceanographic processes, there is no reason to believe the extra food consumption which will arise from a 26% increase in farmed area will do anything but cause a further deterioration in mussel condition.

23. James predicted that mussels on the western side of Beatrix Bay will take ~ 10 weeks longer to grow to 100 mm than they do now if production increases to 6,000 t/a. Clearly mussels must be under stress if their growth rate slows; this is hardly a sign of good health. In my opinion, mussels are already stressed in Beatrix Bay, and therefore so are the natural fauna in this environment.

Attachment 5

STATEMENT OF EVIDENCE

DR SHAW TREVOR MEAD

March 2015

**IN THE ENVIRONMENT COURT
AT BLENHEIM**

ENV-2014-ENV-CHC-34

UNDER

the Resource Management Act 1991

IN THE MATTER

of appeals under S120 of the Act

BETWEEN

DAVIDSON FAMILY TRUST

Applicant

AND

MARLBOROUGH DISTRICT COUNCIL

Respondent

AND

**KENEPURU AND CENTRAL SOUNDS
RESIDENTS' ASSOCIATION INC AND FRIENDS
OF NELSON HAVEN AND TASMAN BAY INC**

Section 274 Parties

STATEMENT OF EVIDENCE BY DR SHAW TREVOR MEAD

March 2015

Counsel: JC Ironside
6 Moore Road
Wakefield
Nelson 7095
Phone:(03) 541 9227
Email: julian@jcironside.nz

Background

1. My name is Shaw Trevor Mead. I am an environmental scientist based at Raglan. I hold BSc and MSc (Hons) degrees from the University of Auckland (School of Biological Sciences), and a PhD degree from the University of Waikato (Earth Sciences). I am currently an environmental scientist and Managing Director at eCoast, which is a marine consulting and research organisation. I have 20 years' experience in marine research and consulting, have 46 peer-reviewed scientific papers, and have solely or jointly produced over 200 technical reports pertaining to coastal oceanography, marine ecology and aquaculture. I have undertaken over twelve hundred research and consulting SCUBA dives around the coast of New Zealand and overseas, and have led many comprehensive field investigations that have addressed metocean, biological and chemical components of the coastal environment. I am affiliated to the New Zealand Coastal Society (IPENZ) and the New Zealand Marine Science Society, and am on the editorial board of the Journal of Coastal Conservation, Planning and Management. I am also technical advisor for the Surfbreak Protection Society (NZ) and Save the Waves Coalition, which mostly entails consideration of marine structures and developments and the impacts they will or have had on high-quality surfing breaks.
2. I have a background in environmental science, coastal oceanography, numerical modelling, marine ecology and aquaculture. I studied for my MSc degree at the University of Auckland's Leigh Marine Laboratory, undertaking subtidal research there from 1994 to 1996 directed at the fertilisation success of sea urchins as a basis for the sustainable management and development of the commercial market. As part of my MSc degree in Environmental Science, I also completed a 4th year law paper in Environmental Law focussed on the RMA (1991) (the subject of my dissertation was the quota management system law review which was under way at the time and ended in the Fisheries Act 1996). The marine ecological components of my Doctorate were directed towards subtidal habitat enhancement of marine structures, while the physical oceanography component was focussed on understanding the effects of coastal bathymetry on wave breaking characteristics using field measurements (bathymetry surveys, aerial photography and GPS positioning of in situ data collection) and hydrodynamic numerical modelling. More recently, I have been involved in a wide range of coastal consulting and research projects that have included the design of coastal structures and developments, and assessments and monitoring of physical and ecological effects of marine construction, coastal erosion control, marine reserves (annual monitoring of benthic communities, fish and lobster, inside and outside Goat Island and Hahie Marine Reserves for the past 10 years), dredging, outfalls, oil industry, aquaculture ventures and various other coastal and estuarine projects that have

included hydrodynamic (waves and currents), sediment transport and dispersion modelling (including contaminants, suspended sediments, freshwater, hypersaline water, nutrients and petro-chemicals).

3. Further to this, with direct relevance to the present case, I am familiar with Beatrix Bay and the Pelorous Sound, and have previously investigated the sustainability of marine farming in Beatrix Bay (Mead et al., 2001), have undertaken an ecological survey of Beatrix Bay (Mead, 2002a), and provided expert evidence pertaining to the impacts of mussel farming on Beatrix Bay (2002b). More recently I provided a desktop summary of the current level of science and understanding of cumulative ecological impacts of mussel farms ring-fencing coastlines such as Beatrix Bay (Mead and Haggitt, 2014), and provided a review of Mr Davidson's and Mr Forrest's evidence with regard to the current application for Resource Consent (Mead, 2014).¹
4. I was also involved in the Board of Inquiry hearing to consider the NZ King Salmon Co. Ltd's plan change requests in the Marlborough Sounds in 2012, specifically with respect to the ecological significance of the proposed sites, the seabed/benthic effects of the depositional footprints of salmon farms, and the wider ecological impacts and cumulative effects of farm waste deposition.
5. I have read the Environment Court's Code of Conduct for Expert Witnesses and agree to comply with the Code when presenting evidence to the Court. I confirm that the matters addressed in this brief of evidence are within my area of expertise. I can confirm that I have not omitted to consider material facts known to me, which might alter or detract from my opinions expressed within this evidence.

Structure of Evidence and Introduction

6. In preparation for the development of my evidence, I have reviewed:
 - a) The application and associated AEE (Appendices 4 and 5);
 - b) Mr Knight's statement of evidence;
 - c) Dr Taylor's statement of evidence;

¹ The reports and evidence cited in this paragraph are attached as Appendices 1-45 of this evidence.

- d) Dr Grange's statement of evidence;
- e) Mr Holland's statement of evidence²;
- f) Dr Stewart's statement of evidence;
- g) Mr Harvey's statement of evidence;
- h) Mr Podjursky's statement of evidence;
- i) Mr Scholefield's statement of evidence;
- j) Mr Clarke's submission in association with salmon farm applications, and;
- k) A variety of relevant literature, as cited in this evidence and associated reports.

7. The first section of my evidence discusses the evidence of Dr Stewart. I am in agreement with Dr Stewart's evidence, which reiterates, and in many instances updates, the concerns that I have with respect to the impacts of mussel farming in Beatrix Bay, cumulative effects and sustainability, which I have previously described in reports and evidence dating back to 2001 (e.g. Appendices 1-5). Some of these areas are expanded on in the first section.
8. The second section of my evidence is a critique of the evidence submitted by Mr Knight, Dr Taylor and Mr Davidson. Since, as stated above, I am in agreement with Dr Stewart's evidence, which includes agreement with some aspects of these experts evidence, the focus is on areas where I am in disagreement with the applicant's experts that have not been covered by Dr Stewart or my previous reports and evidence with respect to Beatrix Bay in 2002 (i.e. Appendices 1-5). That is, where there has been omission of important factors (e.g. assessment of cumulative impacts, hydrodynamic assessment, etc.), and where I believe that there has been misrepresentation of information.
9. In the third section of my evidence, I summarise my opinions with respect to the impacts of the proposed application and the likely cumulative impacts of mussel farming in Beatrix Bay.

² Although Mr Holland refers to sustainability, sustainable growth, sustainable production and a sustainable manner, there is nothing in his evidence to support any of this terminology. Mr Holland's evidence is focussed solely on production of mussels and has no recognition or relevance to sustainable management of natural and physical resources as defined in Part II of the RMA (1991). Therefore, Mr Holland's evidence is not considered further herein.

Dr Stewart's Evidence

10. As stated above, I have reviewed Dr Stewart's evidence and I am in agreement with it. Dr Stewart's evidence reiterates, and in many instances updates, the concerns that I have with respect to the sustainability of mussel farming in Beatrix and that I have previously described in reports and evidence dating back to 2001. Dr Stewart's evidence also identifies additional areas of concern, which I am also in agreement with.

Description of the Marine Environment

11. The physical description of Beatrix Bay has been presented in various reports and evidence, and so is not repeated here, except for likely the most important factor with respect to mussel farming: current speeds and flushing. A fundamental aspect of a good mussel farm location is high current flow. High currents provide a large volume of water to filter food from, reduce impacts on the seabed and mix the local chlorophyll levels (Waite, 1989). Sites with slow currents are more likely to incur benthic impacts and receive less food. Beatrix Bay has very slow currents and a flushing time of over 2 weeks (i.e. it is poorly flushed), i.e. *Beatrix Bay is fundamentally not an optimum location for mussel farms* (Appendix 4 – Mead and Haggitt, 2014).

Analysis of the Likely Amount of Change in Beatrix Bay Due to Mussel Farming

12. This Section 4 of Dr Stewart's evidence, he reiterates the lack of baseline and monitoring data with which to determine change within Beatrix Bay due to mussel farming. Despite concerns about carrying capacity and the impacts of mussel farming on the indigenous ecology since the 1990's (Gibbs *et al.*, 1992), mussel farming in the bay has continued to expand (from 160 ha of mussel farms in 2000 to 304 ha in 2014). Each individual application for resource consent for new farms and farm extensions offshore has been dealt with in isolation, without consideration of cumulative effects. In each case, partly due to dealing with it in isolation, the investigators have stated that there is no need to monitor. How there can be no monitoring of an extractive (i.e. phytoplankton/nutrients) and disturbing (the benthos, hydrodynamics, light penetration, etc.) activity in the marine environment for a development of several hectares is hard to understand, especially when the 304 ha of mussel farms in Beatrix Bay is considered (>15% of the bay area; likely >19% of the seabed directly impacted by deposition). I do not believe it could not occur in any other industry. Whatever the history behind this, the issue is that we are left with a lack of information with which to assess and quantify change.

13. As a way to consider changes and impacts on the wider environment in the absence of monitoring data, Grange (1999) considered the impacts of mussel farms on cockle populations by comparing Oyster Bay (mussel farms present) and Delaware Bay (no mussel farms present). Grange found differences in the population structure of the Oyster Bay cockles (mussel farms) in comparison to the Delaware Inlet cockles (no mussel farms) that prompted him to conclude that the study provided data which may give the very first glimpse that native species are being compromised and that the potential for aquaculture within the existing licenses may already be sufficient to exceed carrying capacity, to the detriment of both the aquaculture industry and native species (Grange, 1997). In Grange's opinion, the granting of additional consents may not be wise until further research had been undertaken.
14. Prompted by Dr Grange's work, a similar study was undertaken in Laverique Bay that is located on the south eastern side of Beatrix Bay (Appendix 3 – Mead, 2002b). A similar population structure was found in Laverique Bay which, following Dr Grange's conclusions, suggested that native species were being compromised by the intensive mussel farming in Beatrix Bay.
15. Investigations into cockle population structure were further presented in Mr Davidson's evidence at the 2014 Hearing for this application, which I reviewed and found that I disagreed with some aspects his interpretation of the results (Appendix 5 – Mead, 2014). Consideration of cockle population densities has been further pursued using data collected by the applicant (Mr Davidson) from 8 sites (4 with mussel farms and 4 without), with the data then supplied to Dr Grange for analysis. Dr Grange concluded that the differences in population structure between sites are likely due to differences between the physical aspects of the sites (substrate, salinity, etc.) and the inherent patchiness of shore communities, rather than being consistent with the proximity to mussels farms. Like Dr Stewart, I have no problem with Dr Grange's interpretation. However, the results provide no conclusive evidence one way or the other with respect to the impacts of intensive mussel farming on indigenous populations. Dr Grange's evidence is not considered further, since the results provide no evidence one way or the other with respect to the impacts of intensive mussel farming on indigenous populations.
16. Dr Stewart has taken a different and more robust approach to assessing the amount of change to benthic communities in Beatrix Bay due to mussel farming. Dr Stewart surveyed community structure at sites that include an impacted site within Beatrix Bay, an un-impacted site within Beatrix Bay, and a control site distant from Beatrix Bay and intensive mussel farms. This represents aspects of the BACI (before/after, control/impact), the method

of monitoring that like Dr Stewart, I advocate as the best way to determine impacts of mussel farms on the marine environment. Unfortunately, there is no long term monitoring of either control or impact sites in the Sounds with which to determine a baseline 'before', and in addition, due to the multiple impacts on the Marlborough Sounds (as described in Dr Stewart's, Dr Taylor's and Mr Davidson's evidence), there has likely been a 'shifting baseline' over time. However, Dr Stewart's investigation compares a control site, a before site (the site of the current application) and a nearby impact site using multivariate analysis.

17. From this investigation, Dr Stewart concludes that in his opinion, the effect of any mussel farm on benthic communities within 100 m of the farm is unlikely to be less than minor, and believes it is highly likely that the communities on the hard substrate inshore of the proposal will be adversely affected. I concur with Dr Stewart's conclusions. Given the higher currents at the applicant's site, the impacts on the inshore hard substrate (i.e. the shallow subtidal rocky reefs that are considered the most un-impacted areas in the Sounds, are often ecologically significant, and are traditionally avoided by mussel farms - Mr Davidson's evidence) will be more extensive than at other sites. The higher currents will lead to a larger footprint outside the farm boundary (these currents are not high enough to transport fine materials away from the site, rather spread 250-400 tonnes of annual deposition per hectare of the farm over a larger area – discussed below).
18. Dr Stewart summarises his findings with respect to the likely amount of change in Beatrix Bay due to mussel farming in his paragraphs 4.39 to 4.44, with which I concur. In addition, the changes to the community, which are considered more than minor for the proposal site, indicate that large and significant impacts have been, and are, occurring when the whole bay is considered (i.e. cumulative effects).

Cumulative Effects and Conclusions

19. Dr Stewart has described the concerns with cumulative effects in Beatrix Bay for a range of factors, including those that I have highlighted in previous reports and evidence since 2001 (Appendices 1-5), which include changes to currents, deposition, nutrient stripping, nutrient recycling, plankton depletion and community change, shading and fouling/biosecurity. These effects are set out in Section 3 of Dr Stewart's evidence and reiterate, and often update, the concerns that I have previously highlighted with respect to the ecological sustainability of mussel farming in Beatrix Bay (Appendices 1-5). I am in agreement with Dr Stewart in these areas of concern.

20. In Section 6 of his evidence, Dr Stewart considers the ecological sustainability of mussel farming and the likely cumulative effects of mussel farming in Beatrix Bay. The various components of the evidence in Section 6 are also included in my previous reports (Appendices 1-4), and have been updated with further relevant information. I concur with Dr Stewart's Section 6.
21. In Section 7, Dr Stewart presents conclusions to his analyses and discusses the best way to sustain the marine ecosystem in Beatrix Bay. I am in agreement with Dr Stewart's conclusions and his evidence supporting a precautionary approach. "A precautionary approach should be taken and the granting of additional consents may not be wise until further research into the impacts of mussel farms on the wider Beatrix Bay ecosystem have been undertaken – that was the case in 2002 (Appendices 2 and 3 – Mead, 2002a, b), and it is still the case today." (Appendix 4 – Mead and Haggitt, 2014). In addition to Dr Stewart's discussion in Section 7, given the results of the various assessments of carrying capacity for the bay (described below) and the known impacts that have already occurred, or are occurring throughout the bay, it is my opinion that reduction of current mussel stocks in Beatrix Bay are required and restoration measures need to be adopted to sustainably manage the marine ecosystem.

Critique of the Evidence Submitted by Mr Knight, Dr Taylor and Mr Davison

22. Similar to Dr Stewart, it is my opinion that the information presented for the site under consideration does not provide sufficient data or reach any robust conclusions about the current state of the communities on nearby substrata, or the likely impacts on communities due to the proposed mussel farm, or the cumulative impacts of mussel farms in Beatrix Bay (in the context of adding another farm). In general, the assessment of effects as described in the evidence presented has been undertaken in isolation, that is without consideration of any other farming activities in Beatrix Bay (i.e. there is no consideration of cumulative impacts and impacts of farming on the wider ecosystem), refers to literature that is not applicable to the application (e.g. Wilson Bay and the Firth of Thames have no physical similarities with Beatrix Bay), and does not present a great deal of the existing information and literature (including ecosystem modelling) that has been generated concerning mussel farming in Beatrix Bay. I have addressed the majority of the concerns that I have in previous reports (Appendices 1-5), which are also addressed Dr Stewart's evidence. Therefore, I have attempted to focus on particular topics that I disagree with, where there has been omission of important factors, and where I believe that there has been misrepresentation of information.

Mr Knight – Carrying Capacity (Production and Ecological)

23. Mr Knight describes how as early as 1992 (i.e. Gibbs *et al.*, 1992), there were concerns in regard to the whole of the Pelorus Sound with respect to sustainable mariculture and carrying capacity. He also defines the maximum carrying capacity as the production carrying capacity, which is the limit at which increasing shellfish aquaculture would start to reduce total production output of shellfish from a region. Mr Knight repeatedly asserts that production carrying capacity is unlikely to have been exceeded in Beatrix Bay. However, no analyses are required to make this statement – if production carrying capacity was even reached and not exceeded, then the bivalve culture replaces the ecological role of zooplankton and the ecosystem essentially collapses down to a nutrient-phytoplankton-culture-detritus system (Gibbs, 2004), i.e. cultured shellfish and phytoplankton are the only living organisms in the system (a situation which can likely never practically be achieved (Jiang and Gibbs, 2005)).
24. More importantly, production carrying capacity is not relevant to sustainability and the RMA (1991); ecological carrying capacity is what must be considered. Mr Knight notes that ecological carrying capacity is likely to be approximately 20% of the production carrying capacity, i.e. the level of culture that could be introduced without significantly changing the major energy fluxes or structure of the food web (Jiang and Gibbs, 2005).
25. However, Mr Knight applied the approach put forward by the Aquaculture Stewardship Guidelines (ASC, 2012) to assess sustainability/carrying capacity of the application. Unfortunately, Mr Knight assessed only the impacts of the current 7.34 ha application on the carrying capacity of a part of Beatrix Bay (the area of influence). This approach is superfluous; I believe that no calculations are required to conclude that one 7.34 ha mussel farm in the 1,960 ha's of Beatrix Bay will not exceed carrying capacity. Therefore, the ASC (2012) assessment has been reapplied to Beatrix Bay, including the whole bay and all of the mussel farms, in order to assess whether or not the ecological carrying capacity of Beatrix Bay has been exceeded.
26. Of note are the findings of Jiang and Gibbs (2005). Their study area was Tasman and Golden Bay, an area open to the Cook Strait and the nutrient upwellings from the West Coast that form eddies which are shed off the tip of Cape Farewell (e.g. Foster and Battaerd, 1985), i.e. an area of significantly higher nutrient and phytoplankton levels than Beatrix Bay. They concluded that 65 t km² of green mussels could be supported by the system without significantly changing the ecosystem structure (i.e. the ecological carrying capacity). Placed in the context of Beatrix Bay, production is reported by Mr Knight to currently be 6,500

tonnes/yr, and the total area of the bay is ~20 km². This equates to 325 t km² of mussel production, suggesting the ecological carrying capacity has very likely been exceeded in Beatrix Bay.

27. Appendix 6 presents the calculations and results of Mr Knight using the ASC (2012) approach to carrying capacity alongside the calculations and results of using the ASC (2012) approach for the whole of Beatrix Bay side-by-side. The values used in the calculation of the whole bay are from Mr Knight's evidence (e.g. 549 M m³ of volume in the Bay, 20 days residence (flushing) time in the bay, 44.46 million mussels in a 7.43 ha farm extrapolated to 304.4 hectares of the existing mussels (note, the Davidson farm was not included, just today's mussel stock), clearance rates, etc.). The full volume of Beatrix Bay has been used for the retention/flushing time. However, this is a conservative approach, since Pelorus Sound and Beatrix Bay are known to be strongly stratified (e.g. James, 2000). Using only the surface mixed layer in this assessment would further reduce the CT/RT ratio.
28. The CT/RT (clearance time to retention time) that result from the ASC (2012) approach is used to assess carrying capacity. If water renewal (RT) is faster than water clearance (CT), (therefore if CT/RT >1) carrying capacity is not expected to be exceeded. Mr Knight's assessment of only the current proposal results in a CT/RT score of 3.05, which as would be expected, indicates that a 7.43 ha mussel farm in Beatrix Bay will not exceed carrying capacity. When the whole bay and all of the existing mussel farms are considered, the CT/RT ratio is found to be 0.0675, that is significantly <1, indicating that cultured bivalves control the ecosystem (i.e. exceed the ecological carrying capacity of Beatrix Bay). ASC (2012) recommends a further assessment to consider whether the carrying capacity has been exceeded if the result of the CT/RT is <1, which considers the ratio of clearing time to primary production time (CT/PPT).
29. Unfortunately the data to assess the PPT are not available for Beatrix Bay. However, given that the result of 0.0675 is greatly <1 (by 2 orders of magnitude), it would likely be a superfluous exercise. There is little likelihood that CT/PPT will be >3 (and so not exceeding carrying capacity) due to algal buffer stocks within the bay. This is because Beatrix Bay is nutrient deficient with low phytoplankton concentration and most often could be considered oligotrophic (discussed below).
30. In cases where CT/RT and CT/PPT criteria are not met, ASC (2012) recommend bay-wide management plans that address the potential cumulative pelagic effects of multiple farms and reduction of regional stocking levels in order to ensure that the ecological carrying capacity is not being exceeded.

31. Alternative methods of considering carrying capacity include Gibbs (2007). Gibbs (2007) describes methods for assessing the sustainability performance of bivalve aquaculture activities. Again, all calculations were made using the values provided within Mr Knight's evidence and values stated within Gibbs (2007). The first indicator is clearance efficiency (CE), which is calculated retention time (RT) / clearance time (CT) (i.e. the reciprocal of the ASC (2012) method). Very low values of this indicator (<0.05) suggests that the culture will not induce significant changes to the pelagic functioning (i.e. is not impacting on ecological carrying capacity). Values greater than 1.0 indicate that the water in the inlet or bay is flushing slower than the water is processed through the bivalve culture. In such cases the bivalves could be expected to regulate phytoplankton abundance, as the water would be filtered by the culture repeatedly before it would be flushed out of the inlet. The score for Beatrix Bay is an extremely high 14.86, which again indicates the cumulative impacts of mussel farming exceed the ecological carrying capacity.
32. Another indicator is regulation ratio (RR) (Gibbs, 2007). This indicator gives a measure of how much control the bivalves have on the algal population within the Bay. Values close to or exceeding 1.0 suggest that the bivalve culture will be controlling the phytoplankton dynamics in the growing region and this implies that there will be costs to competitors relying on phytoplankton (i.e. ecological carrying capacity is being exceeded). The calculated value for Beatrix Bay is 3.23.
33. Based on the results of these analyses, it is my opinion that ecological carrying capacity in Beatrix Bay has already been exceeded, and rather than granting more mussel farm resource consents, stocking levels in the bay should be reduced (as recommended by ASC (2012)) and restoration measures should be developed.

Mr Knight – Nitrogen Budgeting

34. Mr Knight introduces his evidence on nutrient processing by stating how complex these processes are, and the numerous biological processes that influence nutrient concentrations in the water column and the wider ecosystem. However, he then goes on to present an extremely basic summation of inputs and losses/removals to demonstrate that there is an abundance of nitrogen in the Sounds and Beatrix Bay, and so there should be no concerns with respect to the reduction in nitrogen from Beatrix Bay due to the scale of the single mussel farm. As with Mr Knight's calculations pertaining to ecological carrying capacity, I believe that no calculations or evidence are required to reach the opinion that a single 7.34 ha farm in Beatrix Bay will have little impact on nutrient balances. However, nutrient budgets cannot be assessed in the absence of the large area of existing mussel farms within Beatrix

Bay. Cumulative effects must be considered. Mr Knight has again omitted any assessment of cumulative impacts.

35. It is my opinion that this very coarse assessment of nitrogen sources and sinks provides very little value when considering the impacts of mussel farming in Beatrix Bay (and also omits a large nitrogen sink with respect to Nitrogen burial – with 250-400 tonnes of material being deposited per hectare of farm per year (Harstein, 2005; Hartstein and Stevens, 2005), this value is likely to be substantial). Mr Knight's assessment is basically assuming that the Pelorus Sound is just a single well-mixed water body that is uniform in every respect. This is obviously not the case. Pelorus Sound has a great deal of complexity with respect to physical and chemical processes and magnitudes. As previously stated, a fundamental aspect of a good mussel farm location is high current flow, of which there are areas within Pelorus Sound. Regions of slow currents are more likely to incur benthic impacts and receive less food. Beatrix Bay has slow currents and a flushing time of over 2 weeks (i.e. it is poorly flushed), i.e. *Beatrix Bay is fundamentally not an optimum location for mussel farms.*

Mr Knight – Existing Trophic Status and Nutrient Conditions

36. In this section, Mr Knight states that the main concern regarding nutrient removal relates to the process of oligotrophication, which in general terms involves reductions in primary production (and carry-on ecosystem effects) as a function of nutrient removals from shellfish harvesting activities. Oligotrophication is to be avoided in order to ensure primary production is not reduced, which leads to carry-on ecosystem effects. I agree that nutrient inputs from the open ocean, as well as from terrestrial sources, lead to fluctuations in the typical water column characteristics for different trophic states in the Marlborough Sounds (e.g. Zeldis et al., 2008). However, in this instance Mr Knight is referring to the whole of the Marlborough Sounds as 'low-mesotrophic' (i.e. tending towards oligotrophic). The data for Beatrix Bay indicate that it is oligotrophic.
37. In relation to Table 2 of Mr Knight's evidence, chlorophyll concentrations data for Beatrix Bay indicate that levels are regularly <1 mg/m³ for extended periods of time (Appendix 1 – Mead et al., 2001; Knight's Figure 3). Secchi disk measurements for a 6 month period in 2002 found that the disk was observed to depths of up to 14 m and was always >6 m (unpublished data). These data both indicate an oligotrophic classification for Beatrix Bay. Anecdotal evidence from 4 Marlborough Sound residents all report how the waters of Pelorus Sound and Beatrix Bay are now far clearer than they were in the past.

38. Cornelisen (2013) summarises the main effects associated with extractive forms of aquaculture, i.e., mussel and oyster aquaculture, that may lead to cumulative ecological effects on the wider ecosystem, such as oligotrophication (oligotrophic environments offer little in the way of nutrients to sustain life), changes in the abundance and composition of plankton – which may lead to down-stream effects on the food web. It is also suggested that farming of macroalgae could add to the oligotrophic process by removing dissolved nutrients from the water column. This is of concern with respect to the current application which seeks consent to grow *Macrocystis pyrifera*, *Ecklonia radiata*, *Gracilaria*, *Pterocladia lucida* and *Undaria*.

Dr Taylor – Depositional and Benthic Impacts

39. Dr Taylor's evidence is a broad overview of the AEE for this application (Forrest, 2013). I have previously reviewed the AEE and Mr Forrest's evidence, which is included as Appendix 5 (Mead, 2014). I do not agree with Dr Taylor's assessment of depositional and benthic effects, which are based on the assumption of low to moderate water currents, and literature on benthic impacts under mussel farms that are not in Beatrix Bay. In addition, like Mr Knight, Dr Taylor does not assess the cumulative impacts of mussel farming, only the impacts of the single proposed farm. Dr Taylor describes a large range of activities and stressors that have led to the degradation of the Marlborough Sounds, and suggests that because of all of these past impacts it is difficult to determine the cumulative impacts of mussel farming. Cumulative effects are effects that arise over time or in combination with other effects – regardless of the scale, intensity, duration or frequency of the effect (RMA, 1991). Thus, the effects of mussel farming are not meant to be extracted from within all the other impacts, and since Dr Taylor is able to determine the effects of the proposed mussel farm (through the review of Forrest (2013)), then it follows that the cumulative impacts of mussel farms within Beatrix Bay can also be assessed.
40. In the first instance, water velocities measured at the site (for only 20 hrs) were found to be 1-3 cm/s. I agree that these water velocities are low, however, they are larger than adjacent sites in Beatrix Bay, and so will increase the size of the depositional footprint relative to other farms (i.e. beyond the 20-30 m of impact cited in Dr Taylor's evidence). These currents are not strong enough to re-suspend these fine materials (i.e. >10 cm/s), and once they are consolidated on the seabed will require current speeds well above the threshold for the re-suspension and transportation of fine materials. In addition, the ADCP indicates that current directions are not away from the rocky reef (as does the uncalibrated numerical modelling), and current is directed between the 2 arms of the proposed farm, that is back and forth across the rocky reef habitat (Appendix 1 – Mead and Haggitt, 2014). As a result, it is very

likely that the depositional footprint of the farm will cover much of the rocky reef and have more than a minor impact on the ecological community currently inhabiting this site.

41. Secondly, Dr Taylor states that mild enrichment of the soft-sediment habitat is expected beneath the proposed farm as a result of the biodeposits it generates. Similar to the lack of modelling of the hydrodynamic impacts of the farm³, no investigations of the actual enrichment state beneath existing mussel farms in the vicinity have been undertaken, rather literature not specific to Beatrix Bay is relied upon. Investigation of the state of enrichment under marine farms in Beatrix Bay has found that due to the low current speeds in the Bay and the complete lack of wave orbital motion at the seabed (due to the fetch-limited nature of the Sounds resulting in only small short period waves which cannot penetrate to the >30 m depths), that the benthos is highly enriched, the community composition is greatly modified and supports only a few opportunistic species, and the sediment is anoxic (Appendix 2 – Mead, 2002). 250 to 400 tonnes of deposition has been reported to accumulate beneath each hectare of farm per annum (Harstein, 2005; Hartstein and Stevens, 2005), which in Beatrix Bay cannot be re-suspended and transported away.
42. Similar to Mr Knight's assessment, Dr Taylor (Mr Forrest) has not considered the cumulative impacts of these significant changes to Beatrix Bay. In total, changes to the soft substrate likely represent some 19% of the total area in the Bay (Dr Stewart's evidence), which represents a greater than minor impact to Beatrix Bay. In addition, given the state of the sediment under farms, it is very likely that changes in nitrogen recycling are occurring (along with burial and sequestering). These observations suggest modification to nutrient cycling (Barg, 1992), which has the potential to increase the loss of nutrients (harvesting removes nutrients, but a shift to enhanced de-nitrification converts nitrate to N₂ at a higher rate than previously (e.g. Berelson *et al.*, 1998)) and/or a shift in nutrients released (i.e. increase the release of ammonium instead of nitrate). These changes in sediment nutrient recycling in Beatrix Bay are important, because the nitrogen supplied by sediment release is of a similar magnitude to Cook Strait input, and much larger than river flow or mussel excretion inputs (Gibbs *et al.*, 1992). Ross *et al.* (1999) confirmed that sediment nutrient recycling has a strong influence on the dynamics of the Beatrix Bay ecosystem.

Mr Davidson – The Receiving Environment

³ Any other 7.34 ha marine development would certainly require hydrodynamic modelling to consider the impacts at a minimum, which is a relatively simple exercise, including the inclusion of friction and other coefficients and treatments to represent the mussel farm based on the work of Plew (2011) and others

43. In paragraph 18 and 19, Mr Davidson refers to calculations concerning the amount of space taken up by mussel farms in Beatrix Bay. I have assumed that these numbers have been put forward to indicate that there are large areas of the bay that have not been exploited. However, the percentages presented by Mr Davidson are relatively large and demonstrate the potential for large cumulative impacts on the ecological carrying capacity of the bay.
44. Paragraph 19 reads *"The coastline of Beatrix Bay is 25.7 kilometres long. Backbones (surface structures) on the 37 marine farms span approximately 8.5 km of shoreline length (constituting 33% of the total shoreline.) Despite the presence of many mussel farms in Beatrix Bay, the data shows, approximately 85% of Beatrix Bay is not occupied by mussels farms with 33% of the coast having mussel lines directly offshore"*. However, considering Mr Davidson's Figure 1 and the associated additional 15-20% due to the movement of longlines (Dr Stewart's evidence), I believe the assertion that some 67% of the coast is open space absent of mussel farms is a mis-representation. Indeed, re-analysis of the extent to which the mussel farms extend along the coast indicates that some 69% of the coast has mussel farms in front of it, and that only 31% could be considered open and unoccupied by mussel farms. This discrepancy likely arises from measuring the length of the complex coast (25.7 km), rather than the space available between farms. I disagree with this approach, and consider 69% of the coastal margin occupied, which is also in reference to the 'ribbon of farms' around the coast of the bay and the associated impacts on the indigenous ecosystem, as discussed in Appendix 4 (Mead and Haggitt, 2014).

Mr Davidson – Site Selection

45. In this section, as well as in the latter paragraphs of the section the receiving environment (paragraphs 20 and 21), Mr Davidson describes how many historical activities have degraded the Sounds and altered the natural state. In this case, the implication seems to be that the Sounds, and in particular Beatrix Bay, have been degraded by activities for many decades and so further development in the form of intensive aquaculture will have little additional impact, i.e. the area is already highly impacted, so more impacts are acceptable. This is similar to Dr Stewart's statement in his paragraph 7.39 "It has been said that Beatrix Bay is a "Farming" area as though that somehow validates any ecological changes that ensue from marine farming". I strongly disagree with this kind of approach, which is in direct conflict with the Purpose and Principles of the RMA (1991). The opposite approach is required, that of reduction of impacts and restoration, which is being practised elsewhere in New Zealand (e.g. www.harbourcare.co.nz; www.mhrs.org.nz; www.mangawhaiharbourrestoration.co.nz) and worldwide.

46. Of note in this section of Mr Davidson's evidence is his description of the importance of the shallow subtidal rocky habitats in the Pelorus Sound and the wider Marlborough Sounds, how they are in a relatively natural state, have often been recognised as ecologically significant sites, and how mussel farm location should avoid such areas. Contrary to this, the Davidson Trust application bookends a substantial area of this important habitat and will impact it through deposition from the proposed farm (as discussed above).

Summary – Impacts of the Proposed Application and the Likely Cumulative Impacts of Mussel Farming in Beatrix Bay

47. With respect to the current application, I agree with Dr Stewart's conclusion 7.1; "The current absence of any mussel farm at the tip of the headland in northern Beatrix Bay provides a valuable discontinuity in the string of farms around the perimeter of the bay. It is ecologically significant from a marine perspective in that it is the only remaining exposed mainland promontory with a southerly aspect in Beatrix Bay that has not been developed and, as such, provides a high degree of naturalness absent along much of the bay shoreline."
48. In addition, the Davidson Trust application bookends a substantial area of shallow subtidal rocky reef habitat and will negatively impact it through deposition from the proposed farm (as stated by Dr Stewart, these impacts are likely to be more than minor). The shallow subtidal rocky habitats in the Pelorus Sound and the wider Marlborough Sound of importance since they are mostly in a relatively natural state, have often been recognised as ecologically significant sites, and as such, mussel farm location should avoid such areas. Furthermore, the proposed site is a hydrodynamic anomaly due to the mainland promontory, which results in relatively higher currents. Due to conservation of momentum, a mussel farm in this location that reduces current velocities by up to 70% and also redirects currents will have impacts on current flows at great distances away from the proposed site.
49. Finally, the AEE has omitted to consider cumulative effects, which in my opinion are very significant, far greater than minor, and have led to an exceedance of the ecological carrying capacity and consequent detrimental impacts on the indigenous marine communities. Dr Stewart has identified some of these changes in community structure through statistical analysis of data collected at control, impacted and un-impacted sites. Anecdotal evidence presented by local residents of the Pelorus Sound also recount the large changes and reduction in marine life in the Beatrix Bay area since mussel farming has expanded. While it is noted that there have been many different impacts and stressors on the Pelorus Sound marine environment, the tendency to oligotrophy in Beatrix Bay is supported by science and data. The observations of increasingly clearer water are due

entirely to the extensive mussel farming in the bay and their massive filtering capacity (Appendix 6).

50. Like Dr Stewart, I consider that the high likelihood of cumulative effects from mussel farming within Beatrix Bay, and the uncertainty surrounding the nature and extent of those effects, are compelling reasons to justify the refusal of further mussel farming consents in the bay until effects are better quantified and understood.
51. In a nationwide report by the Department of Conservation, the Marlborough Sounds was identified as being of national conservation importance (cited Davidson and Davidson, 1994), and includes a diverse marine environment with habitats ranging from the common-place and typical, through to significant sites that support rare, unique or special species (Davidson *et al.*, 2011). The sustainable management of the Marlborough Sounds is critical. It is my opinion that Beatrix Bay is not being sustainably managed and that the incremental expansion of mussel farms in the bay (and elsewhere in Pelorus Sound) has negatively impacted on the ecological carrying capacity of the area (it has been exceeded) and resulted in large impacts on the indigenous marine flora and fauna.
52. The Beatrix Bay marine ecosystem has reached this state due to the consistent omission of consideration of cumulative effects in AEE's supporting resource consent applications for farms, while the lack of monitoring of these commercial operations has made it difficult to quantify detrimental effects. Concerns over carrying capacity within Beatrix Bay were raised in the late 1990's, and a large volume of science indicated that there was indeed potential that the ecological carrying capacity of the bay was being exceeded with consequent impacts on the wider environment. Mussel farming in Beatrix Bay has doubled since then. It is my opinion that the science providers should be leading and directing the industry towards sustainable management in line with the Purpose and Principles of the RMA (1991).
53. It is my opinion that the MDC should be undertaking restoration of the Marlborough Sounds following decades of negative impacts through a range of activities. In Beatrix Bay, although there are many activities documented that have contributed to its degradation, in recent decades it is my opinion that the ecological carrying capacity of the bay has been exceeded. In my view, the cause of this is the incremental increase in mussel farming that has been allowed to develop within the bay. This has very likely had, and is having, detrimental impacts on the wider environment and indigenous marine species. Restoration and measures to sustainably manage Beatrix Bay will need to include the reduction of stocking levels of mussels within the bay.



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) _____
) Shaw Mead

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Attachment 6

**ASC Bi-Valve Standard
Spreadsheet Calculation**

Aquaculture Stewardship Council Bivalve Standards (2012)

Beatrice Basin Calculations

If retention time (i.e. flushing) is faster than clearance time (i.e. filtration) then the standard is automatically met. If retention time (flushing) is slower than clearance time (filtration) then primary production time must be at least 3 times faster than clearance time for the standard to be met.

Clearance Time Calculation:

Number of Days to Filter Entire Watercolumn (CT)

Beatrice Bay	Crail Bay	Clova Bay	Beatrice Basin	Footnote
3.88	5.42	2.01	4.59	1

Retention Time Calculations :

Estimated Flushing Time of Bay - Days (RT)

Is CT greater than RT using previous Beatrice Bay estimates of RT ?

Alternative RT Calculation per ASC Standard - Average Tidal Change (Metres)

Therefore Average Water Volume Low Tide - Litres

Therefore Average Water Volume High Tide - Litres

Therefore RT Calculated As Per ASC Standard

Is CT greater than RT using ASC Tidal Exchange RT ?

CT/RT Ratios:

CT/RT ratio based on previous estimates of Beatrice Bay RT

CT/RT ratio based on ASC Tidal Exchange RT

Compare CT to Primary Production Time (PPT):

Estimated PPT

Therefore Minimum CT per ASC Standard

Is CT greater than 3 times the PPT time ?

Farm Reduction Factor

Adjusted CT Given Reduced Farm Area

Is CT greater than 3 times the PPT time with reduced farm area ?

20	20	20	20	2
No	No	No	No	
1.7	1.7	1.7	1.7	
546,000,000,000	378,392,000,000	139,629,000,000	1,289,600,000,000	
580,000,000,000	406,000,000,000	152,600,000,000	1,378,000,000,000	
8.97	7.69	6.10	8.17	3
No	No	No	No	
0.194	0.271	0.100	0.229	
0.433	0.705	0.329	0.562	
2	2	2	2	4
6	6	6	6	5
No	No	No	No	
36%	10%	68%	24%	6
6.07	6.02	6.27	6.04	
Yes	Yes	Yes	Yes	

NB - Significantly <1

NB - Significantly < 1

Clearance Time Variables:

Total Surface Area - Hectares

Total Surface Area - Square Metres

Estimated Average Depth - Metres

Cubic Meters Water

Litres of Water

Total Mussel Farm Coverage - Hectares

Estimated Spat Catching area

Reduction for Warp Line Area

Therefore Cultured Occupation Area (Ha)

Beatrice Bay	Crail Bay	Clova Bay	Beatrice Basin
2,000	1,624	763	5,200
20,000,000	16,240,000	7,630,000	52,000,000
29	25	20	27
580,000,000	406,000,000	152,600,000	1,378,000,000
580,000,000,000	406,000,000,000	152,600,000,000	1,378,000,000,000
300	163	171	634
25	25	31	81
12.50%	12.50%	12.50%	12.50%
240.625	120.75	122.5	484

Surface Structures as Percentage of Bay	12.03%	7.44%	16.06%	9.31%	
Length of Backbone per Hectare	1,300	1,300	1,300		7
Depth of Dropper Lines - Metres	15	15	15		8
Number of Dropper Lines per Metre of Backbone	1.14	1.14	1.14		9
Total Metres of Dropper Line per Hectare of Farm	22,159	22,159	22,159		
Number of Mussels per Metre of Dropper Line	140	140	140		10
Total Mussels per Hectare of Farm	3,102,273	3,102,273	3,102,273		
Filtration Rate - Litres per Day	200	200	200		11
Water Filtered Per Day per Hectare - Litres	620,454,545	620,454,545	620,454,545		
Water filtered per day by all farms	149,296,875,000	74,919,886,364	76,005,681,818	300,222,443,182	12

Aquaculture Stewardship Council Bivalve Standards (2012)

Beatrix Basin Calculations

Footnotes:

- 1** Volume of Bay/Water Filtered Per Day - per Clearance Time Variables above.
- 2** Previous studies of Beatrix Bay have estimated the retention / flushing time to be 24 days (Sutton & Hadfield, 1997), and 20 days, varying from 12-27 for spring and neap tides respectively, by Heath (1976). Full exchange (dilution with Pelorus Sound) 31.2 to 42 days (NIWA Biophysical Model Table 3.5 page 47).

$$RT = -1 \times P / \ln (V_l / V_t)$$
Where P is the tidal periodicity, the length of the tidal cycle (e.g. ~0.5 days for semidiurnal tides) V_l is the total volume of the water body at low tide (liters) V_t is the total volume of the water body at high tide.
- 3**
- 4** PPT of 1-2 days under *reasonable* conditions (ASC, 2012) . 1-2 says is very conservative given the oligotrophic state of the Beatrix Basin.
- 5** ASC say if clearance time is faster than flushing time then primary production time must be at least 3 times faster than clearance time .
- 6**
- 7** Double backbone per row, with rows 18 metres apart
- 8** From NIWA biophysical model; this is very conservative; e.g. Knight 2015 indicates 25 m dropper lines.
- 9** Based on 3,750m of dropper per 110m of backbone per NIWA Biophysical
- 10** Average per metre of dropper per NIWA Biophysical Model Page 57.
- 11** *Gibbs* (1992) 14 litres per hour - 336 Lites per day; K Woodford (Lincoln University) 360 Litres per day.
- 12** Total cultured hectares x filter rate per hectare.

Attachment 7

NIWA Review of ASC Standard Bi-valve Spreadsheet Calculations

17 July 2017

Steve Ulrich
Marlborough District Council
PO Box 443
Blenheim 7240

Dear Dr Ulrich

Review of Spreadsheet calculating Retention and Clearance Timescales for the Beatrix Basin

Kenepuru and Central Sounds Residence Associated (KCSRA) prepared a document titled “Reflections on and Solutions to Mussel Farming Planning Issues in the Marlborough Sounds” which includes calculations on the clearance time of Crail Bay, Clova Bay and Beatrix Bay (their appendix 1). Marlborough District Council invited NIWA’s Dr Niall Broekhuizen (and individuals from other organisations) to participate in a discussion about the KCSRA spreadsheet at a recent TAG meeting. To facilitate the TAG discussions, Dr Broekhuizen created a new spreadsheet and presented it at the meeting. Dr Broekhuizen’s spreadsheet was intended only to facilitate verbal discussions during the TAG. Time constraints meant that NIWA was unable to review Dr Broekhuizen’s spreadsheet before the TAG meeting. Dr Broekhuizen’s spreadsheet contained two key elements: (a) a duplicate of the calculations presented by KCSRA and (b) an alternative set of calculations. Those yielded different estimates of clearance time due to differences in parameters used in the calculations.

I have been asked to review the spreadsheet prepared by Dr Broekhuizen, check for errors in the calculations, and comment on the parameters used in his calculations. This letter documents my findings.

Description of the spreadsheet

The spreadsheet provided consists of two pages (henceforth, “Sheet1” and sheet “Map”). “Sheet1” contains calculations and comments. Sheet “Map” shows a map of the bays in question including the farmed areas. Hard-copies of “Sheet1” are presented in appendices 1 & 2 of this letter. In appendix 1, the comments that Dr Broekhuizen chose to insert/associate with some of the cells of the worksheet are displayed. I have added a small number of additional comments – these are also shown. In appendix 2, those comments are hidden – to enable the numerical values etc. to be more readily seen. Appendix 3 of this letter displays the “Map” sheet.

Within “Sheet1”, Dr Broekhuizen has recreated the calculations done by Trevor Offen, which were provided in hard-copy form in their document described above. Columns A-F contain KCSRA calculations and comments/footnotes. Values in these comments have either been directly copied from the hard copy, or in some cases calculated from these values. Dr Broekhuizen has also copied comments and footnotes from the original. I have compared the original (hard copy) and regenerated spreadsheet and find that the values agree other than minor rounding differences (which are insignificant) and formatting. This tells me that any calculation steps added by Dr Broekhuizen are consistent with those used in the original calculation. An additional 2 lines have been added (lines 63 to 64) which are calculations by Dr Broekhuizen to check the longline density (number per hectare) used by KCSRA.

Columns G-L contain calculations conducted by Dr Broekhuizen. The main differences (which will be discussed in more detail below) are in how Dr Broekhuizen has calculated the number of mussels present in each bay.

General comments

I have found no errors in the *calculations* in the spreadsheet.

Bay volumes and Retention Times – I have not checked the volumes used. It appears volumes are calculated from surface area (the extents are not provided by KCSRA) and an estimate of average depth.

It appears that a tidal period of 13 hours has been used, which is longer than the dominant M2 tidal period of 12.42 hours. This would inflate the estimates of retention time by ~ 5%, but I do not consider this significant given the inherent uncertainty in the ASC methodology of calculating retention time.

It should be noted that the ASC method gives a retention *time-scale* – that is an order of magnitude type estimate rather than a number that represents the true retention time of the bays. Consideration should also be given to what the retention time means physically – is it the time to replace all the water, or a time-scale over which a substantial proportion is exchanged? (e.g. 1/e, 90%, 99%?)

Clearance times – implicit in the calculation of clearance time is that the mussels remove all seston from water filtered (i.e. 100% capture), and that there is no refiltration (i.e. water is filtered only once). That last assumption is clearly incorrect – some water will be filtered by mussels multiple times as it moves through a farm or number of farms. Consider also that in a tidal flow, water moves through a farm multiple times before residual currents or dispersion removes it from the farm. Conversely, water in other parts of the bays may never pass through a farm.

In short, this approach of comparing Retention Times and Clearances times gives a guide only, and not a definitive answer to the effect of mussel farms on seston.

Points of agreement

Both sets of workings (Offen, Broekhuizen) have used the same values for number of mussels per length of dropper line. This value (140 per m) is reasonable for harvest size mussels on a well-stocked line. It may overestimate for larger size classes (I have observed ~110 per m for 83mm shell length (Plew, D.R. 2005; Plew, D.R. *et al.* 2009), while Gibbs, M.T. (2007) assumed 150 per m although it is not clear if he considered mixed sizes or harvest size).

Both use a filtration rate of 200 l/day per mussel, which equates to 8.33 l/hr. While KCSRA suggest this is conservative, citing filtration rates of up to 14 l/hr, Dr Broekhuizen indicates he feels this value to be on the high side. Elsewhere, 5 l/hr has been used (Gibbs, M.T. 2007). Ren (NIWA, pers comm) suggests 6-9 l/hr for adult mussels (Plew, D.R. *et al.* 2009). 200 l/d is reasonable, but potentially on the high side.

Points of difference

Bay areas – Dr Broekhuizen has obtained different bay surface areas to KCSRA. The extents of each bay used in Dr Broekhuizen's areas are demarked in the MAP worksheet. KCSRA have not provided a map showing the extents of the bays as used in their calculations. The biggest difference being in Clova Bay

where Broekhuizen's 12.01 km² is ~60% larger than the 7.63 km² used by KCSRA. I assume that KCSRA defined the entrance to Clova Bay as being further into the bay.

Farm areas used by KCSRA and those obtained by Broekhuizen from GIS files provided by Marlborough District Council also differ. Note that Broekhuizen uses KCSRA areas in his calculations, and the other values are provided for information only. However I suggest this be revisited, and a check also made of the volumes of each bay at high and low tide.

Table 1 Comparison of bay surface areas and total farmed areas provided by KCSRA with those obtained by Broekhuizen based on GIS. Note that the areas provided by KCSRA have been used in all calculations.

Bay	Bay area (hectares)		Total farm area (hectares)	
	KCSRA	Broekhuizen	KCSRA	Broekhuizen
Beatrix Bay	2000	1994	300	319
Crail Bay	1624	1372	163	135
Clova Bay	763	1201	171	227
Beatrix Basin	5200	5665	634	681

Different approaches have been used to calculate the number of mussels present in each bay:

KCSRA approach: first the actual cultured occupation area has been calculated by removing areas used for spat catching (25 ha Beatrix, 25 ha Crail, 31 ha Clova Bays respectively), and then reducing the remaining area by a further 12.5% to account for space lost for warp lines/anchoring. I cannot confirm these values (particularly the spat catching area). KCSRA then assume 1300 m backbone per hectare. I believe this is too high. Assuming a double backbone longline (i.e. 2 backbones per longline, or 650 m longline per hectare), this equates to a longline spacing of ~15 m ⁽¹⁾.

I looked at a number of farms throughout the wider Beatrix Basin, and found that longline spacing varied between 17 m and 30 m. The average spacing was closer to 25 m, although 20 m could be taken as a conservative value. I would therefore estimate 400-500 m of longline per hectare of culture area

KCSRA then assume 1.14 droppers per m of backbone and a dropper length of 15 m. Assuming a double backbone longline has a typical length of 110 m, this gives ~2900 m of crop rope per long-line, which is less than the 3500-4000 m typically used (<http://www.marinefarming.co.nz/public/faqs/>). This partly compensates for the narrow longline spacing used in their calculations, but still results in what I believe to be an overestimate of the number of mussels present. KCSRA then calculate the length of crop rope per hectare of farmed area (1300 m backbone/ha x 1.14 droppers/m backbone x 15 m dropper length = 22,230 m/ha). Multiplied by the number of mussels per m length of crop rope (140), filtration rate (200 L/day) and the cultured area (total farm area less spat catching areas and 12.5% warp allowance); this gives the total volume of water filtered per day.

Dr Broekhuizen has calculated a typical number of long-lines per hectare using values from the Marine Farming website (<http://www.marinefarming.co.nz/public/faqs/>). A typical farm has 9 longlines in a 3 ha area, giving 3 longlines per hectare. Each longline supports 3500 to 4000 m of crop rope² (this value is

¹ 1300m/2 = 650 m longline per 100m x100m hectare, or 6.5 x 100m longline lengths per 100m of farm width. 100 m/ 6.5 = 15.4 m spacing

² The calculations use 3750 m per longline, which is the mid-point of this range. Increasing or decreasing the length of crop rope per longline has a linear effect on the clearance time. At 4000 m per longline, the calculated clearance time would be only 7% lower. Similarly, at 3500 m the

supported by C. Johnstone), giving $3 \times 3750 \text{ m} = 11,250 \text{ m}$ crop rope/ha. Dr Broekhuizen then multiplies this crop rope density by the number of mussels per m, the filtration rate and cultured area to derive his estimate of daily volume of water filtered.

While Dr Broekhuizen has used the same cultured area as KCSRA (total mussel farm coverage less spat catching area and warping area), I think that due to the vague definition of farm size on the MFA website, perhaps he should use farm area including the area used for anchoring/warp lines. This would increase the number of mussels and therefore total filtration by 12.5%.

As part of checking the spreadsheet to see the impact of different assumptions used, I have recalculated the clearance time using two different methods to estimate the number of mussels present. Firstly, I followed KCSRA's approach but assumed an average longline spacing of 20 m (which I think is conservative, i.e. at the narrow end of spacings seen in the bays), and 15 m dropper lengths at 2×1.14 droppers/m length of longline as used by KCSRA. This gives 17,045 m of crop rope/ha. When multiplied by the cultured area in each bay, number of mussels/m and filtration rate, this gives total filtration volumes of 114,840,688 m³/d (Beatrix), 57,629,145 m³/d (Crail) and 58,464,350 m³/d (Clova).

The resulting clearance times are 5.05, 7.05 and 2.6 days for Beatrix, Crail and Clova Bays respectively.

My second approach was to use the total length of longlines present in each bay, as established from aerial photographs taken in 2012 (Broekhuizen, N. *et al.* 2015). These lengths are 60,649 m, 36,379 m and 39,667 m in Beatrix Bay, Crail Bay and Clova Bay respectively. A typical longline is 110 m long³, holding on average 3750 m crop rope, or $3750/110 = 34.1 \text{ m}$ crop/m longline. The total filtration rate for each bay is therefore calculated as total length of longlines \times 34.1 m crop rope per m longline \times 140 mussels/m \times 200 l/day. I make an allowance by adding 10% for longlines that may not have been present during the aerial survey. The resulting clearance times are 9.1, 10.7 and 3.7 days for Beatrix, Crail and Clova Bays respectively.

The following table summarises clearance times calculated by KCSRA, Broekhuizen and myself.

Table 2 Comparison of clearance times (in days) using KCSRA farm areas, bay volumes, 140 mussels/m crop rope, 200 l/day/mussel filtration rate.

Method	Beatrix Bay	Crail Bay	Clova Bay	Total Beatrix Basin
1. KCSRA	3.88	5.42	2.01	4.68
2. Broekhuizen (original)	7.65	10.67	3.95	9.21
3. Broekhuizen (using cultured area plus warp area)	6.70	9.34	3.46	8.06
4. Using KCSRA method but increased longline spacing	5.05	7.05	2.61	6.05
5. Using total measured longline length + 10%	9.13	10.66	3.67	9.48

As the table illustrates, there is a range in estimated clearance times due to different assumptions made in determining the number of mussels present in each bay. My opinion is that KCSRA have overestimated the number of mussels present by assuming longlines are spaced closer than they appear to be. Dr Broekhuizen may have underestimated the total number of mussels present by using cultured area excluding the 12.5% warp area – in line 3 of Table 2 I have used total farmed area (excluding spat catching area), which reduces

clearance time would increase by 7%. These differences are minor compared to the differences between clearance times between Broekhuizen and KCSRA.

³ Satellite imagery shows that some longlines in Beatrix Bay are up to 200 m long. I assume here for simplicity that the length of crop rope per m is the same as for the 'normal' 100-120m longlines

clearance time by 12.5% but these values are still higher than KCSRA estimates. If longline spacing is increased to 20 m (which I think is conservative, 25 m would be more typical), I obtain clearance times slightly less than Dr Broekhuizen, but greater than KCSRA. Using the actual length of longlines observed gives clearance times closest to Dr Broekhuizen's original estimates.

To improve on any of the estimates above would require obtaining (most likely from industry) the total length of crop rope used within these systems.

Summary

- I have examined the spreadsheet prepared by Dr Broekhuizen
- No errors were found in the calculations (i.e. the implementation of the equations)
- Dr Broekhuizen's estimates of clearance time are higher than the KCSRA estimates due to different estimates of the total number of mussels present in each bay
- In my opinion, KCSRA estimates of the length of backbone per hectare of farm are too high, leading to a faster (smaller) clearance time than other estimates
- Dr Broekhuizen's estimates of clearance time may be slightly high due to using cultured area only (excluding the anchor/warp area), whereas the longline density he has used is likely based on farm consented area. This would reduce his estimates of clearance time by only 12.5%.
- When I redo the calculations using the length of long-lines observed during 2014 aerial surveys (adding an additional 10% to account for any lines not in the water at the time), assuming 3750 m crop rope per 110 m length of long-line, I obtain clearance rates that are closest to those of Dr Broekhuizen. This seems to me the simplest and likely most accurate estimate of the total number of mussels, and hence clearance rate.
- To improve on these estimates would require obtaining (likely from industry) the total length of crop rope present in these bays
- Note there is uncertainty in the filtration rate per mussel, averaged over 'harvest size' mussels
- There is also uncertainty in the number of harvest size mussels per m length of crop rope
- Bay areas and volumes have not been checked. While all calculations have used the same values for consistency, these will have some impact on retention time estimates.

Yours sincerely

David Plew
Scientist (Hydrodynamics)

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Gibbs, M.T. 2007: Sustainability performance indicators for suspended bivalve aquaculture activities. *Ecological Indicators* 7: 94-107.

Plew, D.R., 2005. The hydrodynamic effects of long-line mussel farms. Ph.D Thesis, University of Canterbury, Christchurch, 328 pp.

Plew, D.R., Enright, M.P., Nokes, R.I. and Dumas, J.K. 2009: Effect of mussel bio-pumping on the drag on and flow around a mussel crop rope. *Aquacultural Engineering* 40: 55-61.

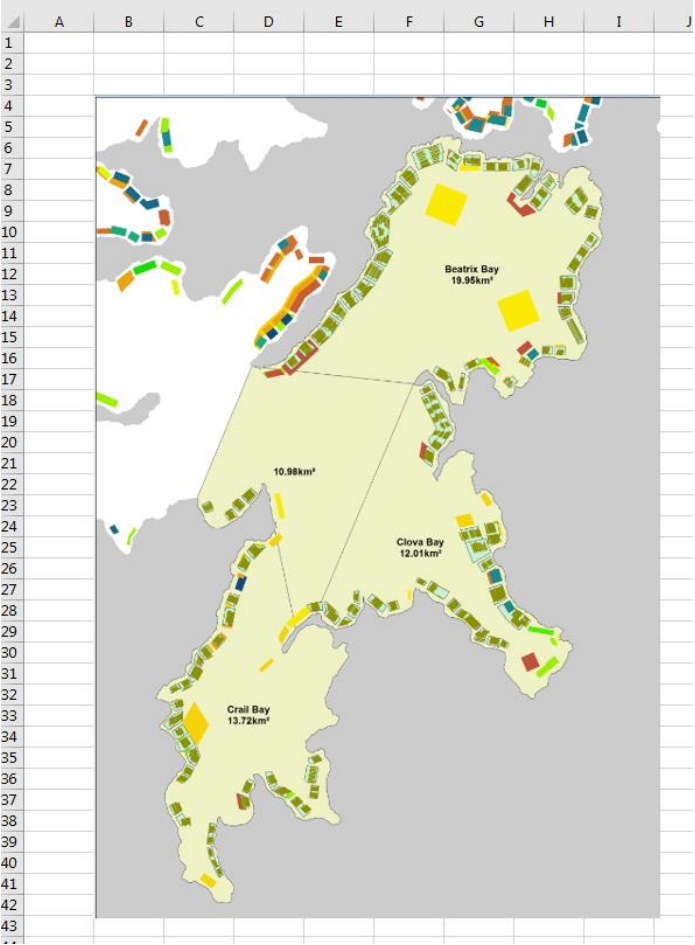
Appendix 1. "Sheet1" with comments displayed

	A	B	C	D	E	F	G	H	I	J	K	L	M	N	O	P	Q	R
1		THESE ARE THE	Niell				THESE ARE NIELL'S VALUES											
2			Brackhuizen:															
3	Aquaculture Stewardship Council Bivalve Standard (2012)		column A, row 3-35, including row 66-70															
4	Beatrix Barin Calculation																	
5	If retention time (i.e. flushing) is faster than clearance time (i.e. filtration) then the standard is automatically met. If retention time (flushing) is slower than clearance time (filtration) then primary production time must be at least as fast as the standard to be met.																	
6																		
7																		
8																		
9																		
10	Clearance Time calculation:	Beatrix Bay	Crail Bay	Clava Bay	Beatrix Barin	Fantasia												
11	Number of Days to Filter Entire Water Column (CT)	3.88		5.42	2.01	4.59	1											
12																		
13	Retention Time Calculation:																	
14																		
15	Estimated Flushing Time of Bay - Days (RT)	20		20	20	20	2											
16	If CT greater than RT using previous Beatrix Bay estimate of RT? No	Na	Na	Na	Na	Na												
17	Alternative RT calculation per ASC standard - Average Tidal Cycle	1.7		1.7	1.7	1.7												
18	Therefore Average Water Volume Lau Tide - Litres	546,000,000,000		375,392,000,000	139,629,000,000	1,285,600,000,000												
19	Therefore Average Water Volume High Tide - Litres	580,000,000,000		406,000,000,000	152,600,000,000	1,379,000,000,000												
20	Therefore RT calculated as per ASC standard	9.97		7.69	6.10	9.17	3											
21	If CT greater than RT using ASC Tidal Exchange RT? No	Na	Na	Na	Na	Na												
22																		
23	CT/RT Ratio:																	
24	CT/RT ratio based on previous estimate of Beatrix Bay	194		0.271	0.101	0.230												
25	CT/RT ratio based on ASC Tidal Exchange RT	433		0.705	0.320	0.562												
26																		
27	Summary CT to Primary Production Time																	
28	Estimate PPT	2		2	2	2	4											
29	Therefore Minimum CT per ASC standard	6		6	6	6	5											
30	If CT greater than 2 times the PPT time? No	Na	Na	Na	Na	Na												
31																		
32	Farm Reduction Factor	0.353333333		0.096666667	0.665	0.235	6											
33	Adjusted CT Given Reduced Farm Area	6		6	6	6												
34	If CT greater than 2 times the PPT time with reduced farm area? No	Na	Na	Na	Na	Na												
35																		
36	Clearance Time Variables:																	
37		Beatrix Bay	Crail Bay	Clava Bay	Beatrix Barin													
38	Total Surface Area - Hectares	2000	1624	763	5200													
39	Total Surface Area - Square Metres	20,000,000	16,240,000	7,630,000	52,000,000													
40	Estimated average Depth - Metres	25	25	25	25													
41	Cubic Metres Water	500,000,000	406,000,000	192,600,000	1,379,000,000													
42	Litres of Water	580,000,000,000	406,000,000,000	192,600,000,000	1,379,000,000,000													
43	Total Muzzle Farm coverage - Hectares	200	163	71	520													
44	Estimated Spot catching area	25	25	25	25													
45	Reduction for Wasp Line Area	12.50%	12.50%	12.50%	12.50%													
46	Therefore Cultured Occupation Area (Ha)	240.425	120.75	12.5	493.075													
47	Surface Structure as Percentage of Bay	12.03%	7.44%	16.06%	9.21%													
48	Length of Backline per Hectare	1,200	1,200	1,200	1,200													
49	Depth of Drapper Line - Metres	15	15	15	15													
50	Number of Drapper Line per Metre of Backline	1.13439	1.13439	1.13439	1.13439													
51	Total Metres of Drapper Line per Hectare	22,159	22,159	22,159	22,159													
52	Number of mussels per metre of drapper	140	140	140	140													
53	Total Mussels per Hectare of Farm	3,102,236	3,102,236	3,102,236	3,102,236													
54	Filtration Rate - Litres per Day per m ²	200	200	200	200													
55	Water Filtered per Day per Hectare - Litres	620,447,100	620,447,100	620,447,100	620,447,100													
56	Water Filtered per day by all farms	149,295,033,428	74,919,987,325	76,004,769,750	300,218,640,943													
57	Time to clear bay (days, Offshore)	3.88		5.42	2.01													
58																		
59	Time to clear bay (days, MFQAQ)																	
60	Implied landline per ha (2 backlines per langline)	5.91	5.91	5.91														
61																		
62																		
63	Niell Brackhuizen: 200/24-3 L/h - around the highest value recorded. That is bold, and inconsistent with 140 mussels per m ² drapper car at that density, mussels unlikely to be large enough to achieve 9 L/h																	
64																		
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Appendix 2: "Sheet1" with comments suppressed

	A	B	C	D	E	F	G	H	I	J	K	L	M	N	O	P	Q	R
1		THESE ARE THE KCRA NUMBERS					THESE ARE NIALL'S VALUES											
2																		
3		Aquaculture Stewardship Council Bivalve Standards (2012)																
4		Beatrix Barin Calculation																
5																		
6		If retention time (i.e. flushing) is faster than clearance time (i.e. filtration) then the standard is automatically met. If retention time (flushing) is slower than clearance time (filtration) then primary production time must be at least 3 times faster than clearance time for the standard to be met.																
7																		
8																		
9		Beatrix Bay	Crail Bay	Clava Bay	Beatrix Barin	Fantaster												
10		Clearance Time calculations:																
11		Number of Days to Filter Entire Water Column (CT)	3.88	5.42	2.01	4.59	1											
12																		
13		Retention Time Calculations:																
14																		
15		Estimated Flushing Time of Bay - Days (RT)	20	20	20	20	2											
16		Is CT greater than RT using previous Beatrix Bay estimate of RT?	Na	Na	Na	Na												
17		Alternative RT calculation per ASC standard - Average Tidal Change	1.7	1.7	1.7	1.7												
18		Therefore Average Water Volume Low Tide - Litres	546,000,000,000	378,392,000,000	129,629,000,000	1,289,600,000,000												
19		Therefore Average Water Volume High Tide - Litres	580,000,000,000	406,000,000,000	152,600,000,000	1,378,000,000,000												
20		Therefore RT calculated as per ASC standard	8.97	7.69	6.10	8.17	3											
21		Is CT greater than RT using ASC Tidal Exchange RT?	Na	Na	Na	Na												
22																		
23		CT/RT Ratios:																
24		CT/RT ratio based on previous estimate of Beatrix Bay RT	0.194	0.271	0.101	0.230												
25		CT/RT ratio based on ASC Tidal Exchange RT	0.433	0.705	0.330	0.562												
26																		
27		Compare CT to Primary Production Time (PPT):																
28		Estimate PPT	2	2	2	2	4											
29		Therefore, Minimum CT per ASC standard	6	6	6	6	5											
30		Is CT greater than 3 times the PPT time?	Na	Na	Na	Na												
31		Farm Reduction Factor	0.3523333333	0.0966666667	0.665	0.235	6											
32		Adjusted CT Given Reduced Farm Area	6	6	6	6												
33		Is CT greater than 3 times the PPT time with reduced farm area?	Yes	Yes	Yes	Yes												
34																		
35		Clearance Time Variables:																
36		Beatrix Bay	Crail Bay	Clava Bay	Beatrix Barin													
37		Total Surface Area - Hectares	2000	1624	763	5200												
38		Total Surface Area - Square Metres	20,000,000	16,240,000	7,630,000	52,000,000												
39		Estimated average Depth - Metres	29	25	20	27												
40		Cubic Metres Water	580,000,000	406,000,000	152,600,000	1,404,000,000												
41		Litres of Water	580,000,000,000	406,000,000,000	152,600,000,000	1,404,000,000,000												
42		Total Muzzle Farm coverage - Hectares	300	163	171	634												
43		Estimated Spot catching area	25	25	31	81												
44		Reduction for Warp Line Area	12.50%	12.50%	12.50%	12.50%												
45		Therefore Cultured Occupation Area (Ha)	240.625	120.75	122.5	483.875												
46		Surface Structures as Percentage of Bay	12.03%	7.44%	16.04%	9.31%												
47		Length of Backbone per Hectare	1,200	1,200	1,200	1,200												
48		Depth of Drapper Liner - Metres	15	15	15	15												
49		Number of Drapper Liner per Metre of Backbone	1,136.35	1,136.35	1,136.35	1,136.35												
50		Total Metres of Drapper Liner per Hectare of Farm	22,159	22,159	22,159	22,159												
51		Number of muzzles per metre of drapper line	140	140	140	140												
52		Total Muzzles per Hectare of Farm	3,102,236	3,102,236	3,102,236	3,102,236												
53		Filtration Rate - Litres per Day per muzzle	200	200	200	200												
54		Water Filtered per Day per Hectare - Litres	620,447,100	620,447,100	620,447,100	620,447,100												
55		Water Filtered per day by all farms	149,295,053,430	74,918,967,325	76,004,764,750	300,210,840,513												
56		Time to clear bay (days, Offen)	3.88	5.42	2.01	4.68												
57		Time to clear bay (days, MFATAO)																
58		Implied laneline per ha (2 backbone per laneline)	5.91	5.91	5.91													
59																		
60		Issues:																
61		Adopted a fairly high filtration rate that appears inconsistent with total mussel density and much of the filtration literature.																
62		Adopted a high estimate for the length of drapper per ha of marine farm area																
63		drapper may not extend to 15m																
64		Offen's estimate of area of Clava Bay is much smaller than mine. Where does he draw the seaward boundary?																
65																		
66		Fantaster																
67		1 Volume of Bay/Water Filtered Per Day - per Clearance Time Variables above.																
68		2 Previous studies of Beatrix Bay have estimated the retention/flushing time to be 24 days (Sutton & Hadfield, 1997), and 20 days, varying from 12-27 days spring and neap tides respectively, by Heath (1976). Full exchange (dilution with Polarur Sound) 21.2 to 42 days (NIWA Biophysical Model Table 3.5 page 47).																
69		3 RT = $\frac{1}{P} \ln \left(\frac{V_H}{V_L} \right)$ Where P is the tidal periodicity, the length of the tidal cycle (e.g. 0.5 days for semi-diurnal tides) V _L is the total volume of the water body at low tide (litres) V _H is the total volume of the water body at high tide.																
70		4 PPT is 1-2 days under reasonable conditions (ASC, 2012). 1-2 days is very conservative given the oligotrophic status of the Beatrix Barin.																
71		5 ASC says if clearance time is faster than flushing time then primary production time must be at least 3 times faster than clearance time.																
72		6 Percentage reduction in cultured farm area required to meet the ASC standard of CT > 3 times PPT from																
73		7 Double backbone per row, with rows 18 metres apart																
74		8 From NIWA biophysical model; this is very conservative; e.g. Knight 2015 indicates 25m drapper liner.																
75		9 Based on 3,750m of drapper per 110m of backbone per NIWA Biophysical model page 57.																
76		10 Average per metre of drapper per NIWA Biophysical Model Page 57.																
77		11 Gibbs (1992) 14 litres per hour - 336 litres per day; K Woodford (Lincoln University) 360 litres per day.																
78		12 Total cultured hectares x filter rate per hectare.																

Appendix 3: Sheet “Map”



Attachment 8

NIWA Biophysical Model for the Pelorus Sound

June 2015

A biophysical model for the Marlborough Sounds

Part 2: Pelorus Sound

Prepared for Marlborough District Council

June 2015

Prepared by:

Niall Broekhuizen
Mark Hadfield
David Plew

For any information regarding this report please contact:

David Plew
Hydrodynamics Scientist
Hydrodynamics
+64-3-343 7801
david.plew@niwa.co.nz

National Institute of Water & Atmospheric Research Ltd
PO Box 8602
Riccarton
Christchurch 8011

Phone +64 3 348 8987

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Reviewed by



Graham Rickard

Approved for release by



Charles Pearson

Formatting checked by



David Plew

Executive summary

The Marlborough District Council contracted NIWA to undertake biophysical modelling of the Queen Charlotte and Pelorus Sounds. The purpose of the modelling was to describe the effects of existing and proposed mussel and fin-fish farms on water quality. This report presents results for Pelorus Sound. Results for Queen Charlotte Sound and Tory Channel were presented in a previous report.

The biophysical model consists of a three-dimensional hydrodynamic model (with 20 layers in the vertical) coupled to a biogeochemical model (which models water quality, plankton, and other biological and chemical attributes). We used the ROMS hydrodynamic model with the Fennel biogeochemical model, with additional components added to simulate mussel and fish farms. The biogeochemical model includes: (a) the inorganic nutrients ammonium and nitrate, (b) a single phytoplankton class, (c) a single zooplankton class and (d) three classes of particulate organic detritus (slow and fast sinking natural detritus and very fast sinking organic detritus stemming from mussel and fish farms (mussel faeces and pseudo-faeces, fish faeces and waste food)). The abundances of most of these are characterized by means of nitrogen concentration, but the phytoplankton is characterized by two variables: nitrogen concentration and chlorophyll concentration.

A total of seven farming/biogeochemical scenarios were modelled:

- No mussel or fish farms (with benthic denitrification operating) [henceforth, NM-NF-WD].
- Existing mussel farms (no fish farms, with benthic denitrification operating) [EM-NF-WD].
- No mussels, existing fish farms with benthic denitrification [NM-EF-WD].
- Present day/existing farms scenario (with benthic denitrification operating): mussel farms in operation in 2012 (counted by aerial-surveys), and New Zealand King Salmon Ltd. Salmon farms that operated during 2012/2013 (Waihinu Bay, Forsythe Bay, and two farms in Crail Bay). [EM-EF-WD]. We were instructed to treat results from this scenario as a 'baseline' against which to compare alternative scenarios.
- *Approved farms*: as for the *present day* scenario, but also including the additional mussel and fin fish farms that have been approved or existed but were not occupied during the 2012 aerial survey [AM-AF-WD]. The additional salmon farms are Richmond, Waitata and Port Ligar¹. We also include a small farm in Beatrix Bay licenced for hapuku, although we assume the feed schedules and physiology are the same as for salmon.
- Existing mussel farms, no fish farms, without benthic denitrification [EM-NF-ND].
- Existing mussel and fish farms without benthic denitrification [EM-EF-ND].

In the *with denitrification* scenarios it is assumed that 75% of any particulate organic nitrogen (from any source) which settles to the bed will be lost from the system through denitrification (whilst the remaining 25% is returned to the water column as ammonium). In the *without denitrification*

¹ The Port Ligar farm was included in error. The licence for a salmon farm at that site has been rejected.

scenarios, none of the sedimenting particulate organic nitrogen is lost from the system. It is all returned to the water column as ammonium.

Simulations spanned 500 days (24 May 2012 to 6 October 2013), consisting of a 135 day spin-up period followed by 365 days (1 year) over which the model outputs were analysed.

Horizontal grid resolutions from 100 m to 400 m were tested. Finer resolution grids provide greater detail of the spatial distributions of both physical (hydrodynamic) and biogeochemical properties, but the simulations take significantly more time to run (halving the size between grid points increases the computation time by a factor of approximately 8). The 200 m model reproduces the essential aspects of the hydrodynamics of Pelorus Sound with acceptable accuracy and allows simulations with the full biophysical model for periods of over one year. The 200 m resolution grid was used when making the biophysical simulations reported within this document.

The hydrodynamic model was compared to current meter data collected from a variety of locations and periods during 1994–1995 and 1997–1998, and for shorter durations (FRIA assessments) during 2005. Temperature and salinity were compared with monthly profiles collected by Marlborough District Council at 11 stations from 2012–2013.

Analysis of the hydrodynamic model output allows us to make the following conclusions about the physical behaviour of the Sound.

- Peak tidal flows through the Waitata Reach vary from 20–30,000 m³ s⁻¹ at neap tide to 50–60,000 m³ s⁻¹ at spring tide.
- However, movement of nutrients and tracers through Pelorus Sound is driven primarily by estuarine circulation. The dominant supply of freshwater is from the Pelorus River.
- The estuarine circulation involves a flow of approximately 5000 m³ s⁻¹ of brackish water at the surface out from Pelorus Sound into Cook Strait, and a similar inflow of ocean water below.
- Sustained low river flows cause the estuarine circulation to weaken, leading to longer residence times within the Pelorus Sound. However the estuarine circulation is seldom, if ever, entirely absent.
- Surface salinities decrease (the water becomes fresher) as one moves from outer to inner Pelorus Sound, but the influence of surface freshening events from increased river flow occurs through the Sound.
- Stratification in Pelorus Sound is generally driven by salinity. In summer time, when river flows are generally low, warmer surface temperatures can strengthen stratification. In winter, surface salinities can be sufficiently low to allow the surface water to become cooler than that of deeper waters.
- The biophysical model was validated using field data collected from 7 stations in Pelorus Sound by Marlborough District Council. We used model coefficients derived during calibration against 3 years of data when we modelled the Queen Charlotte Sound. The comparison between the modelled state variables (NO₃-N, NH₄-N, chlorophyll, phytoplankton nitrogen, zooplankton, small and large detritus) suggests that the model reproduces the majority of the respective long-term averages and the

respective amplitudes of the seasonal cycles moderately well, but frequently fails to reproduce the phase of the seasonal cycle accurately. For example, the onset of springtime phytoplankton growth (and associated reductions of NO_3 concentrations) is late relative to the field data. Similarly, during late summer/early autumn the simulated NO_3 concentrations start to rise later than the field measurements suggest they should. In comparison with the field data, the model appears to over-predict the summertime concentrations of phytoplankton carbon biomass and of chlorophyll. We believe that the apparent quantum of over-prediction is deceptively large. In the model, the phytoplankton and chlorophyll state-variables represent the entire phytoplankton community. In contrast, the field determinations of phytoplankton biomass and chlorophyll used techniques that would have counted only those members of the total phytoplankton community that have cell sizes greater than approximately $2\text{ }\mu\text{m}$. Other studies suggest that phytoplankton $<2\text{ }\mu\text{m}$ represented an average of 29% (~5-65%) of the total phytoplankton community in the upper 15 m of the water of Beatrix Bay. We have made no attempt to calibrate the model to the field-data from Pelorus.

- In some cases there are no direct analogues in the field data for modelled state variables, thus we need to infer their values.
- Because the model has only one phytoplankton class, it has no ability to mimic seasonal changes in phytoplankton community structure.
- Our Cook Strait boundary conditions are based upon scarce field data (monthly measurements at only two depths).
- The insolation intensities that are applied are not corrected for possible seasonal-scale variations in cloud-cover or seasonal and hour-by-hour variations in topographic shade (though the latter will be significant only in narrow parts of the Pelorus system).
- The hydrodynamic model produces summer-time water temperatures which are a bit too low. Since phytoplankton and zooplankton physiology is temperature dependent, this (or possibly incorrect parameterisation of the temperature dependence) could have subtle influences upon emergent population growth rates and standing stocks.
- The wind-fields driving the model derive from wind models that have relatively coarse spatial resolution compared to the width of the channels in Pelorus Sound. In combination with the steep topography, this implies that wind-driven surface-flows and mixing may not be well represented in the hydrodynamic model.

In the context of this modelling, nitrogen release from fish farms, and mussel-farm induced nitrogen transformations (and net removal) are the key mechanisms by which farming might influence water quality. If the model did not reproduce the annual averages and amplitudes, its ability to describe the influence of farms would necessarily be called into question. Fortunately, whilst the model is not accurately reproducing the phases of some seasonal cycles, it is reproducing the annual averages and the amplitudes of the seasonal cycles fairly well. In particular, it reproduces the switch from winter-time light limited phytoplankton growth to summertime nutrient (nitrogen) limited growth. Thus, we believe that the model is performing sufficiently well that it can plausibly predict the magnitude of changes induced by the different scenarios.

Our biophysical modelling is aimed at understanding the influences that mussel farming and fish farming have upon nutrient dynamics and the plankton community. Indeed, at the outset, the focus was upon fish-farming. There are important differences between mussel farming and fish farming. Fish farms rely upon adding feed into the environment. Mussel farms require no feed input. Instead, the crop draws feed out of the environment. Fish farms are net sources of readily bio-available nutrient (much of the nutrient added in feed passes into the environment). Mussel farms are a net sink for nutrient (a small fraction of the nutrient within the particulate organic detritus which the mussels are assumed to consume is eventually harvested). Nonetheless, much of the consumed nutrient is recycled into the environment as ammonium and as faeces and pseudo-faeces. This is a natural part of the mussel growth processes. The mussels serve to: (a) convert living plankton into living mussel flesh, (c) ammonium, and (d) dead particulate organic detritus. They serve to convert dead particulate organic detritus into: (a) living mussel flesh, (b) ammonium, (c) recycled/recreated dead particulate organic detritus. Since some of the ingested food is subsequently released back into the environment as ammonium, mussels can serve to speed the transformation of particulate organic nitrogen back into a dissolved form that is readily consumed by phytoplankton.

Based on the output from the model, we infer that with respect to the ecological and water quality responses of Pelorus Sound:

- Phytoplankton growth tends to be limited by low light intensities and short day-length during the winter months. During the summer months, it tends to be limited by a scarcity of nutrient (nitrogen). As a result of this difference, some of the effects of mussel and fish-farming differ between winter and summer months. For this reason, we often draw a distinction between winter- and summer- periods when summarising the simulated effects of mussel and/or fish farms in the following bullets².
- Relative to the nominated baseline scenario (EM-EF-WD), a no mussel, existing fish with denitrification simulation (NM-EF-WD) yields:
 - Winter-time: lower concentrations of ammonium and nitrate but higher concentrations of particulate organic detritus (dead plankton etc.), phytoplankton and zooplankton. The largest changes in relative concentration are seen in Kenepuru Sound and the largest relative concentration changes are within the zooplankton. There, time-averaged near-surface winter-time seston³ concentrations in the NM-EF-WD simulation are more than double those of the EM-EF-WD scenario (for zooplankton in Kenepuru, substantially more than double). The Beatrix/Crail/Clova system also exhibits similar (but smaller) changes.
 - Summertime: lower concentrations of ammonium, nitrate, higher concentrations of detritus and zooplankton, but phytoplankton concentrations which are similar to (or lower than) those of the EM-EF-WD scenario. During summer, mussels convert particulate organic nitrogen (not directly exploitable by phytoplankton) to ammonium (directly exploitable by phytoplankton). Phytoplankton growth is normally nutrient limited during this time, but in the immediate vicinity of the

² Note also, that in this summary, we focus upon inferences drawn from the *with denitrification* simulations.

³ Collectively, phytoplankton, zooplankton and other small particulate material are referred to as seston. The mussels feed upon phytoplankton, zooplankton and detritus. They release detritus (as faeces and pseudo-faeces). Fish also generate faeces. None of this faecal and pseudo-faecal material is part of the seston because they sink very rapidly whereas, by definition, seston is supposed to be approximately neutrally buoyant.

mussel farms, phytoplankton (which survive passage through the farms) find a plentiful ammonium supply. This enables them to grow quickly – more than offsetting the losses that the population suffered to mussel grazing (the ‘excess accrued phytoplankton biomass being fuelled out of the detritus that was consumed). Once again, the largest changes are in Kenepuru Sound.

- Relative to the nominated baseline scenario (EM-EF-WD), a with mussel, no fish with denitrification simulation (EM-NF-WD) yields:
 - Winter-time: lower ammonium, nitrate and natural⁴ detritus concentrations. With the exception of ammonium, the concentrations differ by less than approximately 1%. Phytoplankton and zooplankton concentrations that are almost identical to those of the EM-EF-WD scenario.
 - Summer-time: lower ammonium, nitrate, natural detritus, phytoplankton and zooplankton. The largest changes (declines in the absence of fish farms) are in Crail Bay (reflecting the presence of licensed farms in Crail Bay and Beatrix Bay and the slower flushing time of these bays in comparison with Waitata reach (which also harbours an existing fish farm at Waihinu Bay). Within Beatrix/Crail/Clova, time-averaged summertime phytoplankton concentration is predicted to be up to about 10% lower in the absence of fish farms. Zooplankton concentration is predicted to be up to about 15% lower.
- Turning to a comparison of the approved farms scenarios (AM-AF-WD) with the baseline (EM-EF-WD), the model predicts that the relatively few additional mussel farms present in the ‘approved farms’ scenarios (over and above those of the ‘existing farms’ scenario) induce water-quality changes that extend out to about bay-scale but amount to only a few percent of the simulated baseline (existing farms) concentrations. Changes are evident in nutrient (esp. ammonium) and seston concentrations. The changes include: increased ammonium concentrations in the vicinity of the farms and depressed concentrations of particulate organic detritus and zooplankton. During the winter, phytoplankton concentrations are slightly depressed by the additional mussel farms. During the summer, they are depressed in the immediate vicinity of the new mussel farms but can become slightly elevated further afield.. The changes induced by these additional farms amount to a few percent of background concentrations. These are small relative to natural variability. For example, during winter, mussel grazing is predicted to induce local depletion of up to approximately 10% relative to the background/baseline (existing farms) simulation. In contrast, field data suggest that the extrema of phytoplankton population biomass can vary three or more fold over the course of a year. Indeed, it can sometimes fluctuate by almost that much over time-scales of weeks and space scales of km or less.
- The model predicts that effects induced by additional fish farming will extend through the entire Pelorus system. The effects upon nutrients are more localised⁵ (and, therefore, more intense) than the effects upon phytoplankton, zooplankton or natural

⁴ The small and large detritus classes of the model that receive dead plankton etc *cf* the XL-detritus class that receives faeces and pseudo-faeces from the mussels and fish.

⁵ Because there are few fish-farms in total. In contrast, there are many mussel farms distributed through much of the Pelorus system

detritus. Relative to the present-day, the modelling suggests that the approved additional fish and mussel farms will induce winter-time phytoplankton biomass changes of <5% and summer-time changes of <15% at most⁶. In winter, phytoplankton biomass will increase slightly in the main channel of central and inner Pelorus but decline within Crail/Clova/Beatrix Bays. In summer, it will increase throughout Pelorus. The greatest (but still relatively small) changes will be in the vicinities of the new fish farms (i.e., in Beatrix/Crail/Clova Bays, and around Richmond/Waitata/Port Ligar).

- Wintertime light limitation acts as a 'bottleneck' which limits the response of short-lived organisms to the increased nutrient concentrations.
- Nutrient inputs associated with the additional fish farms are predicted to increase summertime near-surface phytoplankton standing stocks by 5–10% relative to the existing conditions. The simulated phytoplankton concentrations are higher than is the norm for New Zealand coastal waters, but they would not be higher than values that are intermittently recorded in our coastal waters. That said, the field data indicate that the 'existing conditions' simulation may be over-estimating summertime near-surface phytoplankton abundance⁷. We believe that this underlying (possible) over-estimation implies that the 'additional fish farms' scenario will also contain this embedded tendency to over-estimate.
- Even if the real-world summertime phytoplankton concentrations were to reach those predicted by the model, they would probably not be high enough to begin to change the perceived colour of the water. Nor would they be sufficiently high (for long enough and over sufficiently large areas) for the system to be classified as eutrophic.
- At the whole of Pelorus scale, the majority of the farm derived nitrogen is predicted to be lost through denitrification at the seabed of the Pelorus system rather than by export to Cook Strait.

Deposition footprints of the 8 farm sites were predicted with a particle-tracking model. The deposition modelling shows that deposition of waste from each farm is highest immediately beneath that farm. Footprints beneath farms located in low velocity areas (Beatrix, Crail, Forsyth, Port Ligar, Waihinau) extend only a short distance, but in higher current areas (Waitata, Richmond), footprints are predicted to extend several hundred metres from the farm perimeters in the along-shore direction (although the intensity of deposition at these distances is very low). Historical data from existing NZKS farms indicate that Benthic Enrichment Scores of around 5 are increasingly likely to be exceeded when deposition rates exceed 5–10 kg solids m⁻² y⁻¹. The deposition modelling suggests that few, if any, of the new farms will be able to operate at their maximum consented annual feed loads without breaching agreed benthic standards. With that in mind, it is worth noting that the consent conditions for the new farms require that each farm be developed in a staged manner (contingent upon meeting the agreed environmental standards). Thus, there are safe-guards in place that will help to prevent excessive degradation of the seabed around the farms.

⁶ Excluding the XL-detritus class. Uneaten fish feed and fish faeces passes into this class. Inevitably, when a new fish farm is added, the relative concentration of this material increases dramatically in the vicinity of the new farm.

⁷ The degree of over-estimation may not be as large as one would infer from a simple visual comparison of field data and model: the modelled quantity is total phytoplankton, but the sampling by Marlborough District Council does not capture the smallest phytoplankton which can contribute a substantial fraction of the total phytoplankton community biomass.

1 Introduction

1.1 Background

The Marlborough District Council is the regional authority overseeing the Marlborough Sounds, where approximately 80% of New Zealand's aquaculture production occurs. The majority of the area used for aquaculture is occupied by mussel farms, however there are also a small number of salmon farms, particularly in Queen Charlotte Sound. At the commencement of this project, applications had been submitted to the Environmental Protection Authority for additional finfish farm sites. In light of these applications and the possibility of future proposals to expand finfish aquaculture, the Marlborough District Council (MDC) desired an improved range of tools to enable it to predict ecological implications with more certainty. MDC commissioned NIWA to undertake biophysical modelling of the Queen Charlotte and Pelorus Sounds in order to help it understand potential effects of future aquaculture developments. The information provided from the modelling will be used to help plan for longer term, and identify both risks and opportunities.

The primary motivation for the biophysical modelling is to assess the influence of aquaculture. In the later part of the project, the Marlborough District Council has expressed interest in whether the models can be used for assessing the effects of other activities, such as catchment land-use changes. This is possible (indeed the models incorporate freshwater flows and nutrients from rivers and runoff) but modelling the effects of land-use change or changes in nutrient loads from activities other than aquaculture is outside the scope of the current project.

The two sounds (Queen Charlotte and Pelorus) are modelled separately in this project. This report describes the results for the Pelorus Sound. Results for Queen Charlotte Sound were presented in a previous report: A biophysical model for the Marlborough Sounds, Part 1: Queen Charlotte Sound and Tory Channel (Hadfield, Broekhuizen, Plew 2014a).

1.2 Definition of a biophysical model

In this report, we use the term “biophysical model” to describe a numerical (computer) model that couples physical (hydrodynamic) processes with biological and chemical processes.

The biophysical model is comprised of several component sub-models.

- The ROMS (Regional Ocean Model) hydrodynamic model, which simulates the physical behaviour of water including currents, salinity and temperature.
- A nutrient/phytoplankton/zooplankton/organic detritus (NPZD) model. The particular model that we have adopted includes a simple description of the benthic mineralisation of deposited detritus. For that reason, we will refer to it as the *biogeochemical model*.
- A mussel farm model which focuses upon feeding, respiration and excretion.
- A fish farm model which also focuses upon feeding, respiration and excretion.

The four sub-models are implemented within a single code-base and we will refer to the collective implementation as the *biophysical model*. The biogeochemical model component relies on predictions of transport by water currents by the hydrodynamic model, thus the accuracy of the biogeochemical modelling component depends greatly on the hydrodynamic model adequately

capturing the physical behaviour of the region to be modelled. The hydrodynamics affect the biogeochemical modelling, but we do not allow for the presence of mussel and fish farms to influence hydrodynamics since we believe any such feedback will be negligible at the regional scale. Consequently, this report first focuses on describing the performance of the hydrodynamic model before considering the biogeochemical predictions of the complete biophysical model.

1.3 Scope of this project

The scope of this project is to

1. Conduct 3D hydrodynamic simulations of the Queen Charlotte Sound and Pelorus Sound that accurately simulate tidal, wind-driven and residual currents; and model the changes in stratification over seasonal and annual time periods.
2. Couple the hydrodynamic model with a water quality/biogeochemical model to simulate:
 - The influence of present day aquaculture activities on nutrient concentrations, phytoplankton and zooplankton. This *existing conditions* or *present day* scenario contains (a) those mussel farms which were shown to have backbones in the water during aerial-survey operations flown in 2012; and (b) salmon farms that were licensed to operate during the 2012/2013 period. For Pelorus Sound these are the two salmon farms in Crail Bay, and the farms in Waihinu and Forsythe Bays.
 - A future scenario considering additional mussel and salmon farms that have been approved or existed but were not occupied at the time of the 2012 aerial survey. This is referred to as the *approved farms* scenario. The additional salmon farms are Richmond, Waitata and Port Ligar. We also include a small farm in Beatrix Bay which is licenced for hapuku, although we assume that feed schedules and physiology are the same as for salmon.
 - A *worst case* scenario, which is the same as for *approved farms* but with benthic denitrification processes turned off (such that all particulate organic nitrogen, from any source, which settles to the seabed is returned to the bottom-most layer of the water-column as ammonium).
3. Simulate the deposition of waste matter (faeces) emanating from the fish farms.

In addition to the scenarios described above, we also ran simulations with:

- no mussel or fish farms
- existing mussel farms but no fish farms
- no mussel or fish farms with denitrification turned off.

While these scenarios were not required under the agreed scope, we included them as they provide useful information on the relative effects of denitrification processes in the model; and also of mussel and fish farms both with respect to each other, and also to the background (no marine farms) conditions of Pelorus Sound.

1.4 Outline of this report

In the following sections of this report, we describe

- Section 2: The hydrodynamic model, the area modelled, and the data used as input to this model.
- Section 3: Results from the hydrodynamic modelling, including a comparison to observed data.
- Section 4: The biogeochemical components of the biophysical model and its parameterisation.
- Section 5: Results from the biophysical modelling. Specifically, the following results are presented:
 - *No farms with benthic denitrification.*
 - *Existing mussel farms only with benthic denitrification.*
 - *Existing mussel and fish farms with benthic denitrification.*
 - *All approved mussel and fish farms with denitrification.*
 - *Existing mussel farms without benthic denitrification.*
 - *All approved mussel and fish farms without denitrification.*
 - *No mussel farms, existing fish farms with denitrification⁸*
- Section 6: A discussion of the performance, limitations, and implications of the biophysical modelling.
- Section 7: A description of and results from the deposition model used to simulate finfish farm benthic footprints.

⁸ This scenario was not required in the contract. We ran it for our own edification and offer it in that spirit but the readers should note that this scenario was run on a 400 m resolution spatial grid whereas all of the other ones were run on 200 m grids.

2 Hydrodynamic model: Methods

2.1 Model description

The hydrodynamic model used in this project was ROMS (Haidvogel, Arango et al. 2008), a widely accepted ocean/coastal model. ROMS has a number of optional sub-models, including several alternative biological models. The current project uses the Fennel biological model as described in Section 4.1.

ROMS is a fully 3 dimensional model and is able to simulate the currents forced by tides and wind, as well as the effects of density differences caused by variations in temperature and salinity. In the vertical, ROMS uses a terrain-following coordinate system, i.e., a fixed number of levels (here 20) is fitted between the bottom and the surface; this system is well suited to coastal situations and copes well with large tidal variations in sea level. In the horizontal, ROMS uses a structured rectangular (as used in this project) or curvilinear grid. There are several aspects of the ROMS structure that relate to its suitability for the present application.

1. The ROMS grid cannot be fitted around complicated coastlines: instead land is represented by masking out grid cells. This leads to some inefficiency, because in the model grids used for this project less than 50% of the area is occupied by water.
2. The horizontal spacing of a ROMS grid cannot be reduced for better resolution in specific areas, e.g., around a fish farm or in a small bay.
3. ROMS uses a time splitting scheme for the equations of motion, i.e., it solves for the depth-average velocity on a short time step and for the vertical variations from that depth average on a longer time step. For the 200 m simulations described here the short time step was 1.5 s and the long-time step was 12 s. The time-splitting scheme is computationally efficient when the maximum depth in the model domain is large (a few hundred metres or more) but has no advantage in shallower water.
4. ROMS uses an explicit time-stepping scheme, which means that the time step is constrained to a maximum that depends on the grid spacing and the flow speed.

2.2 Model grids and bathymetry

The Pelorus Sound model domain (i.e., the area over which the calculations are performed) is shown in Figure 2-1. It was designed to cover the Sound, plus the area immediately outside in Cook Strait. The domain axes were rotated by 40° anticlockwise from true north/east to better align the domain with the Sound. The exact placement of the boundaries was fine-tuned to avoid instabilities caused by the strong Cook Strait tidal currents interacting with the topography near the boundary.

The model bathymetry was constructed from a number of sources, including:

- a digital terrain model of Marlborough Sounds at 25 m resolution generated from NIWA bathymetry data
- contour data (most digitised from LINZ hydrographic charts) held in the NIWA marine bathymetry database
- high-resolution coastline data (to fix the zero contour in the model), and

- land elevation data (also to improve interpolation near the coast).

These data were collated and interpolated onto the model grid with the GMT⁹ mapping tools. A terrain-following model like ROMS requires a further smoothing process to limit the steepness of the bathymetry. At the 200 m and 100 m resolutions this does not degrade the accuracy of the bathymetry significantly, e.g., at 100 m resolution (Figure 2-1) the model captures sharp features like the pair of dramatic, scour-induced depressions near Cape Jackson.

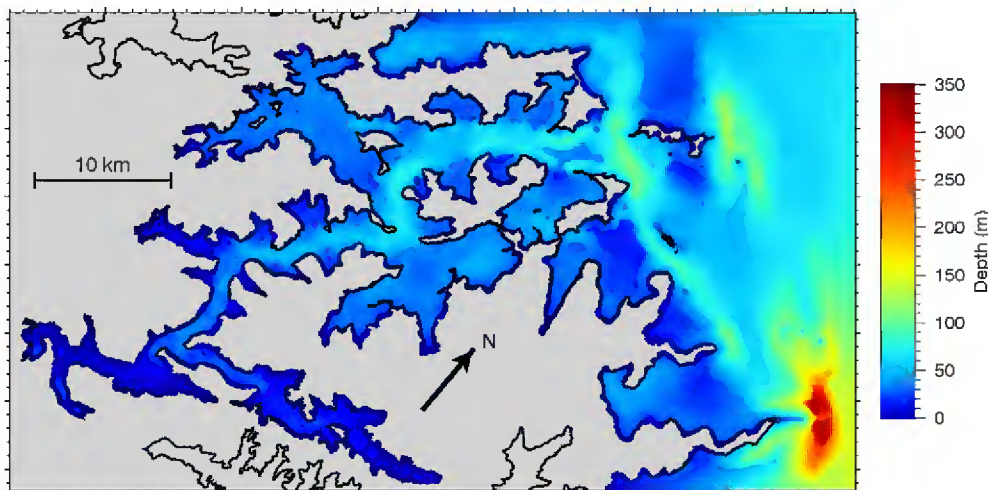


Figure 2-1: The Pelorus Sound model domain and bathymetry. A map showing the model bathymetry and land mask (100 m grid) with LINZ coastline data (black). Note that while parts of the neighbouring Croisilles Harbour and Queen Charlotte Sound are within the domain, these regions were blanked out (shown as grey in the above figures) and were not modelled here.

2.3 Hydrodynamic model simulations

The majority of model simulations described in this report have been for a period of 500 days (24 May 2012 to 6 October 2013) which allows for 135 days to spin-up various components of the model (notably the biogeochemical model) followed by 365 days over which the model output is analysed. Given the short model time steps that are required in coastal situations, a simulation of this duration can be very expensive computationally. Running the model on finer resolution grids allows spatial variability in both physical and biological properties to be better represented, but this comes at the cost of the model taking longer to execute. There is a balance to strike between sufficiently fine resolution and manageable execution time. To examine this issue we set up a series of model grids on the same domain, with different horizontal grid spacing. We employed three such grids: 400 m, 200 m and 100 m.

The execution time (Table 2-1) increases by a factor of approximately 8 with each halving of the resolution, except that between 400 m and 200 m the factor is somewhat smaller because the 400 m grid is not large enough to use the computer efficiently. The addition of biogeochemical processes with the mussel and fish farm parameterisations results in a very large (~ 5–6 times) increase in execution time. This results in part from the extra tracers and processes that the model has to handle, but mainly because the introduction of a fast-sinking detritus class (Section 4.1) required a

⁹ <http://gmt.soest.hawaii.edu/>

change in the model's advection scheme (i.e., the set of model code that moves material through the model grid). With the biogeochemical and aquaculture options activated, the model used the ROMS implementation of the MPDATA advection scheme (Margolin and Smolarkiewicz 1998), which guarantees that concentrations can never become negative. For the hydrodynamics and tracer-flushing simulations we used the usual ROMS third-order upwind scheme (Shchepetkin and McWilliams 1998), which allows small negative concentrations: this is normally acceptable, but leads to problems when the model deals with fast-sinking material. The MPDATA scheme increases execution times by a factor of 2–3.

Table 2-1: Grid resolution and execution time. Time required to execute the Pelorus Sound model for 500 days at three different grid resolutions on a single node of the NIWA supercomputer, Fitzroy. Values in italics have been estimated by extrapolation.

Execution time (days)	400 m	200 m	100 m
Hydrodynamics only	0.6	2.8	25.0
Hydrodynamics plus 6 tracers	0.8	4.2	<i>37.5</i>
Hydrodynamics plus biogeochemical model, mussel farms and fish farms	3.0	17.7	<i>158.0</i>

The 500-day period (24 May 2012 to 6 October 2013) described above was used for the biogeochemical simulations. In addition several hydrodynamics-only simulations were carried out for earlier periods in which hydrodynamic field data were available (Section 2.5):

- 20 May 1994 to 24 June 1995, bracketing a period of measurements in Beatrix Bay
- 24 May 1997 to 28 June 1998, bracketing a period of measurements in Beatrix Bay and outer Pelorus Sound
- 8 July 2004 to 12 August 2005, bracketing a period of measurements for a FRIA (Fisheries Resource Impact Assessment) programme in Pelorus Sound.

2.4 Initial and boundary conditions

The simulations described in this report were all carried out in forward mode, i.e., the model's temperature, salinity, velocity, sea surface height and biogeochemical variables were set to a plausible initial state and then stepped forward in time subject to various forcings from the surface (wind stress, heat flux, freshwater fluxes), the bottom (bottom drag), the open ocean lateral boundaries (specified temperature, salinity, velocity, etc.) and inflows from point sources like rivers. There was no process of adjustment towards observations during the model run (i.e., data assimilation), as there would be in a forecasting model, for instance.

The initial and boundary data for the hydrodynamic variables were taken from a larger-scale model covering Cook Strait (Figure 2-2) at a resolution of 2 km. The Cook Strait model was run for the same periods as the Pelorus Sound simulations, with model fields saved as consecutive six-hour averages. These data were then interpolated to the boundaries of the Pelorus Sound model and written to data files that were read by the latter model. This process is known as one-way, off-line nesting.

The Cook Strait model itself required lateral boundary data. For the 2004–2005 and 2012–2013 simulations this was taken from a global ocean analysis and prediction system operated by the US

Naval Research Laboratory, using the HYCOM¹⁰ ocean model. (The specific dataset used here is the HYCOM/NCODA Global 1/12° Analysis on grid GLBa08.) The HYCOM system provides daily snapshots of the three-dimensional state of the global ocean on a 1/12° grid; at NIWA we have archived a subset of this data around New Zealand since 2003. For the 1994–1995 and 1997–1998 periods, lateral boundary data for the Cook Strait simulations was taken from an implementation of ROMS for the New Zealand region, forced by six-hourly surface fluxes, essentially repeating the work of Rickard, Hadfield and Roberts (2005). In either case the purpose of the Cook Strait model was to generate realistic temperature, salinity and currents at the entrance to Pelorus Sound.

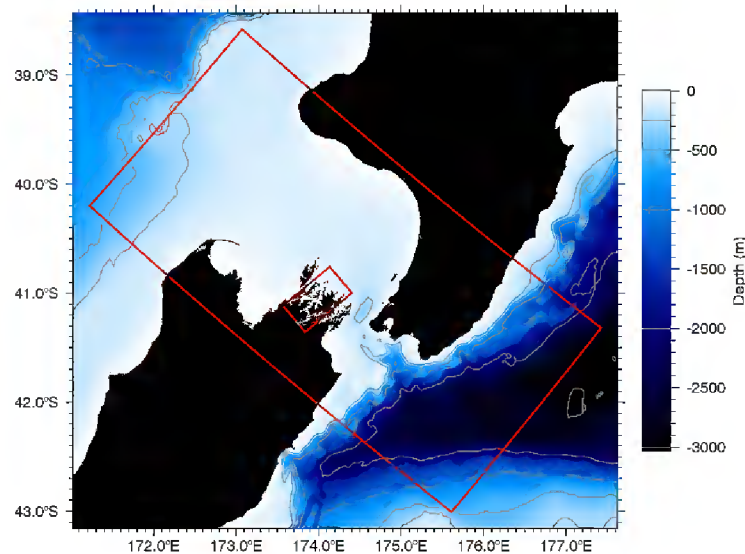


Figure 2-2: The Pelorus Sound and Cook Strait model boundaries.

In principle, the Cook Strait model could include tidal fluctuations in sea-surface height and velocity, which would then be passed into the Pelorus Sound model through its lateral boundaries. However this would require the outer model data to be saved at intervals of ~30 minutes, which would require very large output files. Therefore tides were not represented in the Cook Strait model but were applied at the boundaries of the Pelorus model. Amplitude and phase data for 13 tidal constituents (M2, S2, N2, K2, K1, O1, P1, Q1, 2N2, MU2, NU2, L2, T2) were interpolated from the output of the NIWA New Zealand region tidal model (Walters, Goring, Bell 2001). The ROMS tidal forcing scheme then calculated tidal sea surface height and depth-averaged velocity at each time step and added them at the boundaries.

Surface stresses generated by the wind are an important factor in forcing currents in Cook Strait and (we expect) in Pelorus Sound. For the 2012–2013 simulation, these stresses were calculated using 3-hourly winds from the NZLAM 12 km regional atmospheric model¹¹. For the 2004–2005 and earlier simulations the winds came from a global (1/4°, six-hourly) ocean surface wind dataset called CCMP (Atlas, Hoffman et al. 2010). In both cases, the surface stress was calculated from the wind speed using the following equation.

¹⁰ <http://hycom.org/>

¹¹ NZLAM is part of the NIWA Ecoconnect environmental forecasting system: <http://EcoConnect.niwa.co.nz/>

Equation 2-1: Formula for calculating model surface stress from wind speed

$$\tau = \rho_{air} C_D U_h^2$$

where τ is the stress, ρ_{air} the density of the air, U_h the wind speed and C_D a wind-speed-dependent term called the drag coefficient (Smith 1988). For the larger Cook Strait model, it was found in a previous modelling exercise (Hadfield 2013) that the modelled currents agreed well with measurements, but only when the drag coefficient was multiplied by a factor of 1.4. A similar adjustment—though often by a smaller factor—has been found to be necessary in coastal modelling exercises around New Zealand by us (Hadfield and Zeldis 2012) and others (e.g., P. McComb *pers. comm.*). For the Pelorus Sound model the drag coefficient was not increased as the speed of the modelled wind-driven currents matches the measurements reasonably well without any adjustment. The relatively coarse spatial resolution of the wind datasets means that they will not reproduce the topographic channelling of the wind that is seen in Marlborough Sounds and this can be expected to limit the accuracy of the hydrodynamic model. It is possible to run an atmospheric model at much finer resolution to generate more detailed wind fields, but this is outside of the scope of the current work. We note that from mid-2014 NIWA have an atmospheric model running at 1.5 km resolution which may give improved results in the Marlborough Sounds compared to the 12 km model. Unfortunately this could not be used for the present study but could be considered in the future.

Surface heat fluxes in both the Cook Strait and Pelorus Sound models were calculated using data (6-hourly averages) from a global atmospheric analysis system called the NCEP Reanalysis (Kalnay, Kanamitsu et al. 1996), with a heat flux correction term that causes the model sea surface temperature (SST) to be nudged towards observed SST (the NOAA Optimum Interpolation 1/4° daily SST dataset (Reynolds, Smith et al. 2007)). The heat flux correction prevents the modelled SST from departing too far from reality due to any biases in the surface fluxes, but has a negligible effect on day-to-day variability.

The surface freshwater flux (precipitation minus evaporation) was calculated from a combination of NCEP Reanalysis 6-hourly evaporation data and daily rainfall from the Crail Bay climatological station (NIWA Climate Database¹² agent number 4232). The average annual rainfall at Crail Bay is 1675 mm. Applied over the area of Pelorus Sound (inside the boundaries shown in Figure 3-12 below) of 382 km², this implies a mean rainfall input of 6.4×10^8 m³ per year, or 20.3 m³ s⁻¹.

¹² <http://cliflo.niwa.co.nz/>

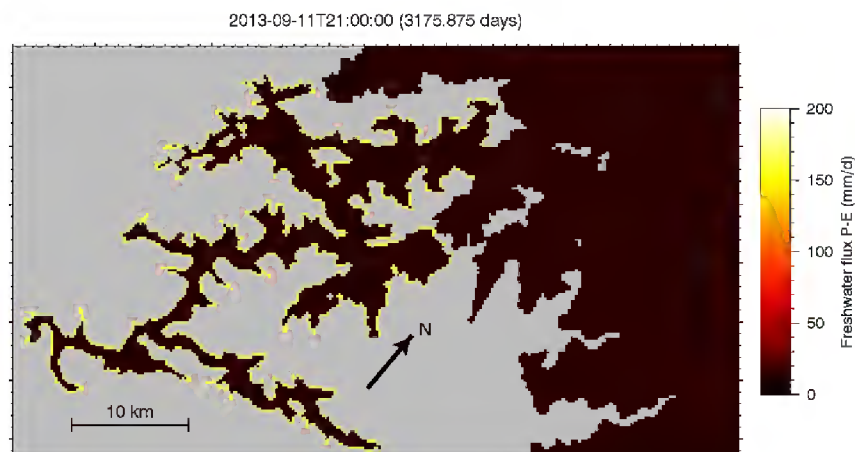


Figure 2-3: Surface freshwater flux. A colour plot showing a snapshot of the surface freshwater flux on the 200 m grid during a moderate rain event ($\sim 10 \text{ mm d}^{-1}$), illustrating the extra input of freshwater in a band next to the coast in Pelorus Sound.

The two largest rivers draining into Pelorus Sound (Heath 1974) are the Pelorus River (catchment area 880 km^2 , annual mean flow $43 \text{ m}^3 \text{ s}^{-1}$) and the Kaituna River (catchment area 115 km^2 , annual mean flow $5 \text{ m}^3 \text{ s}^{-1}$). These two rivers were represented explicitly as point sources in the model. The flow rate for the Pelorus River was constructed from daily average flow data from the NIWA hydrological database (Kathy Walter *pers. comm.*) at the Pelorus at Bryants (58902) and Rai at Rai Falls (58903) stations, with the sum multiplied by 1.2 to account for contributions from catchments downstream of the stations. There is no suitable gauge data for the Kaituna River, so river input to the model was constructed by multiplying the Pelorus River flow by a factor of $5/43$. In addition to the largest rivers, there are many smaller rivers and streams, almost all ungauged, draining into Pelorus Sound. We assumed that the catchment area of Pelorus Sound (excluding the Pelorus and Kaituna River catchments and the surface of the sound itself) is 1075 km^2 (Heath 1974) and that, of the rainfall falling on that area every day, 20% is lost to evaporation and the remainder is immediately delivered to the sea at the coastline. This was achieved by applying an increment to the surface freshwater flux (i.e., an extra input of freshwater, see Figure 2-3) of an appropriate amount in all model grid cells that are adjacent to the land mask and inside the boundaries of Pelorus Sound. The annual mean input by this mechanism is $18.0 \times 10^8 \text{ m}^3$ per year, or $45.7 \text{ m}^3 \text{ s}^{-1}$.

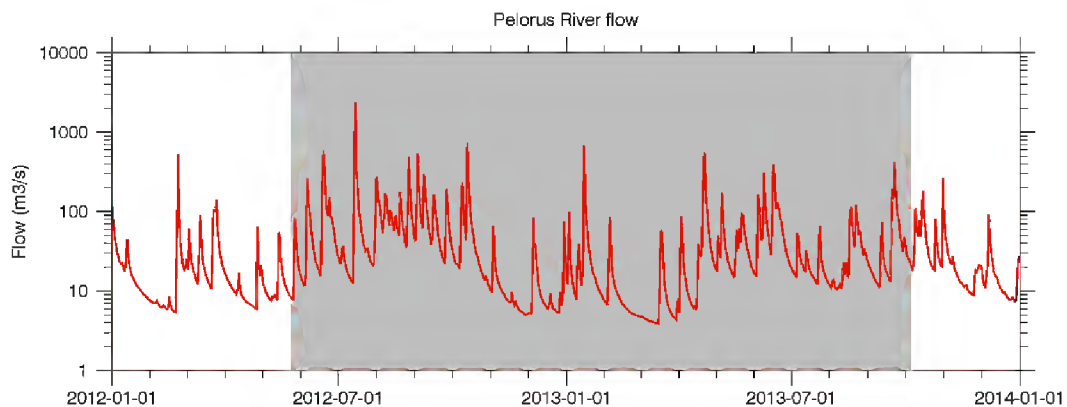


Figure 2-4: Pelorus River flow. Pelorus River flow as input to the model, estimated from the Pelorus at Bryants (58902) and Rai at Rai Falls (58903) stations. The grey rectangle indicates the period of the 500-day 2012–2013 simulation.

2.5 Hydrodynamic field data

2.5.1 Pelorus Sound 1994–1995 and 1997–1998 measurements

Two hydrodynamic measurement campaigns were conducted by NIWA in the 1990s. The first involved a tide gauge and several moorings with current meters and temperature and salinity sensors in Beatrix Bay from September 1994 to June 1995. Selected results were written up by Sutton and Hadfield (1997). The second involved similar instruments deployed in Beatrix Bay and outer Pelorus Sound from July 1997 to February 1998. Current and sea level data from several measurement sites (Figure 2-5) are compared with model output in Sections 3.2.1, 3.3.1 and 3.4.1.

2.5.2 FRIA measurements, 2005

FRIA (Fisheries Resource Impact Assessment) was a programme assessing the effect of aquaculture for the Ministry of Fisheries, a predecessor to the Ministry for Primary Industry. To support these assessments, NIWA and other organisations made hydrodynamic measurements in several aquaculture areas around New Zealand. The Pelorus Sound FRIA measurements were made in February and March 2005: some 19 sites in inner and central Pelorus Sound were occupied by acoustic Doppler current profiler (ADCP) instruments for periods typically between 14 and 28 days. These sites are shown in Figure 2-6 along with an extra site in Port Ligar, labelled as number 20, which was occupied for 22 days in April–May 2005. ADCP pressure and velocity data are compared with hydrodynamic model output in Sections 3.2.2 and 3.3.2.

Permission to use the FRIA Pelorus Sound ADCP data for the present report was kindly granted by the Marine Farming Association¹³.

¹³ <http://www.marinefarming.co.nz/>

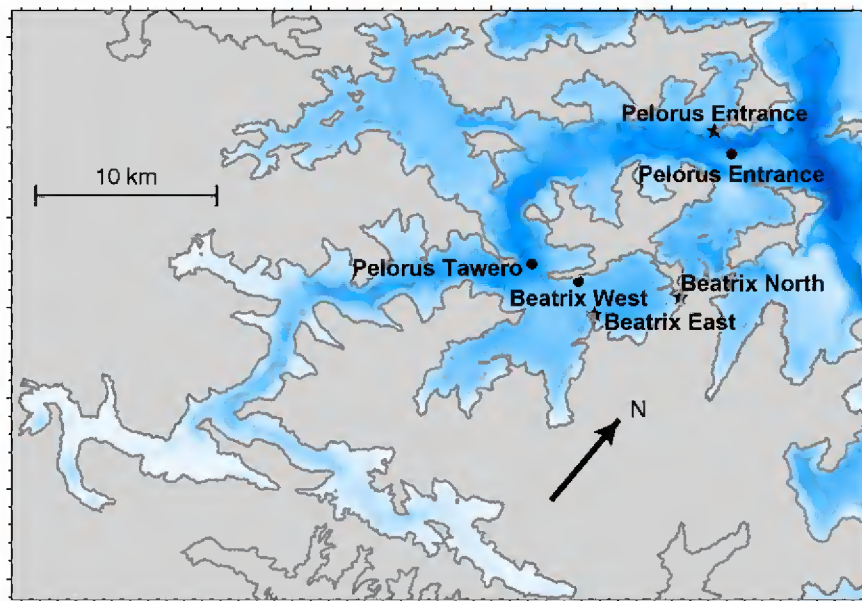


Figure 2-5: Pelorus Sound 1994–1995 and 1997–1998 measurement sites. A map showing the model bathymetry and land mask (100 m grid), with current meter sites indicated by filled circles and tide gauge sites indicated by a star.

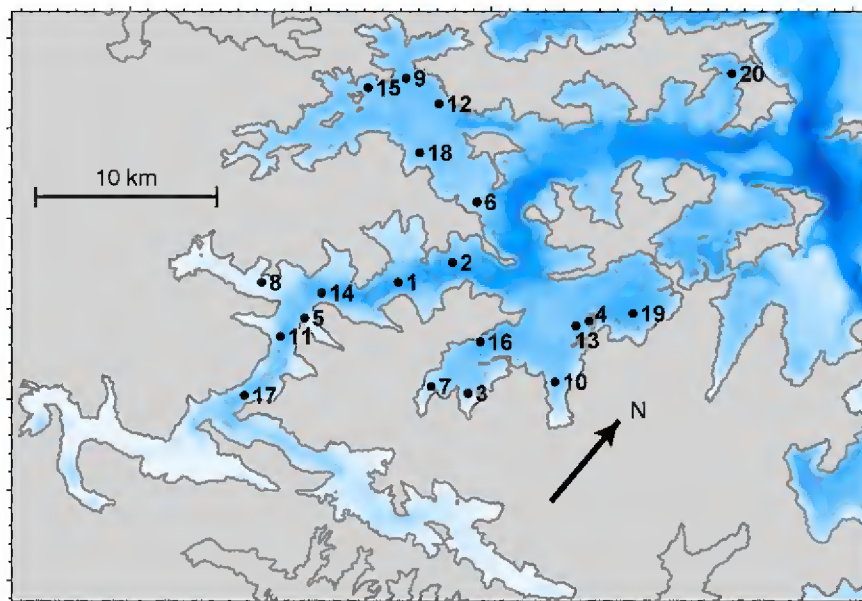


Figure 2-6: Pelorus Sound FRIA sites. A map showing the model bathymetry and land mask (100 m grid), with ADCP sites for the 2005 FRIA project.

2.5.3 Havelock tide gauge data

Sea level time series data for the Havelock tide gauge from 21 April to 14 December 2009 has been kindly supplied by Glen Rowe of Land Information New Zealand, with permission from Port Marlborough. Tidal analyses of these data are compared with model output in Section 3.2.3.

2.5.4 Pelorus Sound CTD surveys, 2012–2014

Beginning in July 2012, Marlborough District Council with NIWA support has measured monthly vertical profiles of temperature and salinity with a CTD (conductivity-temperature-depth) instrument at 11 sites (Figure 2-7) in Pelorus Sound. (At 7 of these sites water quality samples were also collected, see Section 4.4.) The CTD data are compared with hydrodynamic model output in Section 3.1.1.

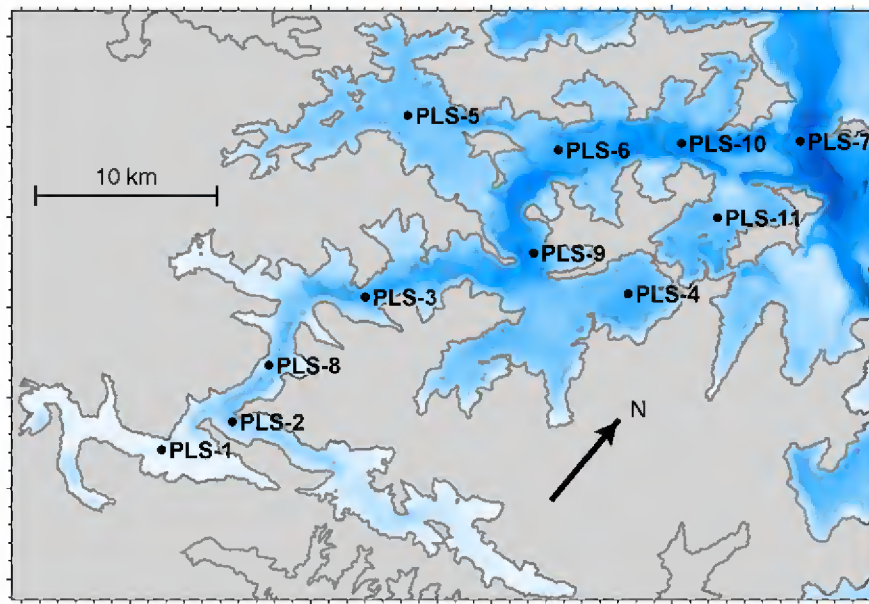


Figure 2-7: Pelorus Sound CTD sites. A map showing the model bathymetry and land mask (100 m grid), with sites occupied by the Pelorus Sound CTD programme. Water quality measurements were also made at seven of these sites.

3 Hydrodynamic model: Results

3.1 Model vs observations: temperature and salinity

3.1.1 Pelorus Sound CTD surveys, 2012–2014

Figure 3-1 shows the temperatures measured by the monthly CTD surveys as colour plots against time and depth axes, along with comparable model data. (However note that the way the CTD data are graphed suggests the temperature is uniform throughout the month, but in fact it only applies to a period of an hour or so, and there is considerable within-month variability in the actual temperatures, just as there is in the model.) The panels of Figure 3-1 show 5 sites from inner to outer Pelorus Sound: PLS-1 (Moetapu Bay) in Mahakipawa Arm, PLS-3 (Yncyca Bay) in Popoure Reach; PLS-4 (Beatrix Bay); PLS-5 (Dart Rock) in western Tawhitinui Reach; and PLS-10 (Post Office Point) in Waitata Reach. Site locations are shown in Figure 2-7.

At all the sites there is a clear seasonal variation in near-surface temperature, from 10–12 °C in late winter to 18–20 °C in late summer. At the sites in the inner Sound the winter minimum is lower and the summer maximum higher than in the outer Sound: the time series plots below will show this more quantitatively. The depth profile of temperature is nearly uniform in late winter: perhaps a little cooler at the surface than at the bottom, but again the time series plots show this better. The warming in spring is confined to the top 10 m or so of the water column, but as summer progresses this warm layer thickens and eventually occupies the full depth, down to the 40 m shown in the figures.

Overall, Figure 3-1 suggests that the variation of temperature with depth and time agrees well between the model and the CTD measurements, with the obvious limitation that the CTD data are monthly snapshots.

Figure 3-2 is a comparison of the temperature measured by the CTD with co-located model data—like Figure 3-1 above—but in this case the data are plotted as time series from two depths, 5 and 30 m. At the innermost and shallowest site, PLS-1 (Figure 3-2a), where there are no 30 m data, the 5 m modelled temperature agrees with the CTD data (bearing in mind the limitations of the CTD data). At all other sites there is good agreement between the model and the measurements in winter and early spring, but from late spring to late summer, the model is biased low by 1–1.5 °C. However the difference in temperature between the two depths remains about right.

The model's temperature bias in summer in Pelorus Sound is thought to be a result of the amplitude of the seasonal cycle in SST in Cook Strait being too low. This might be a result of a bias in the surface heat flux formulation (which is based on coarse-resolution data from a global-scale model) or maybe excessive tidal mixing in the areas with high tidal current speeds in Cook Strait (see Figure 3-8 below).

A noticeable feature of the temperature time series at the four deeper sites in Figure 3-2 is that the near-surface is warmer than the water below in summer, but cooler by as much as 1–2 °C in winter. Given that water expands as it warms, a lower surface temperature can only be maintained if the surface water is less saline, and the salinity data presented below confirm that this is the case. This phenomenon of a cool surface water layer in winter was noted in Beatrix Bay by Sutton and Hadfield (1997) and appears to be a ubiquitous feature in Pelorus Sound.

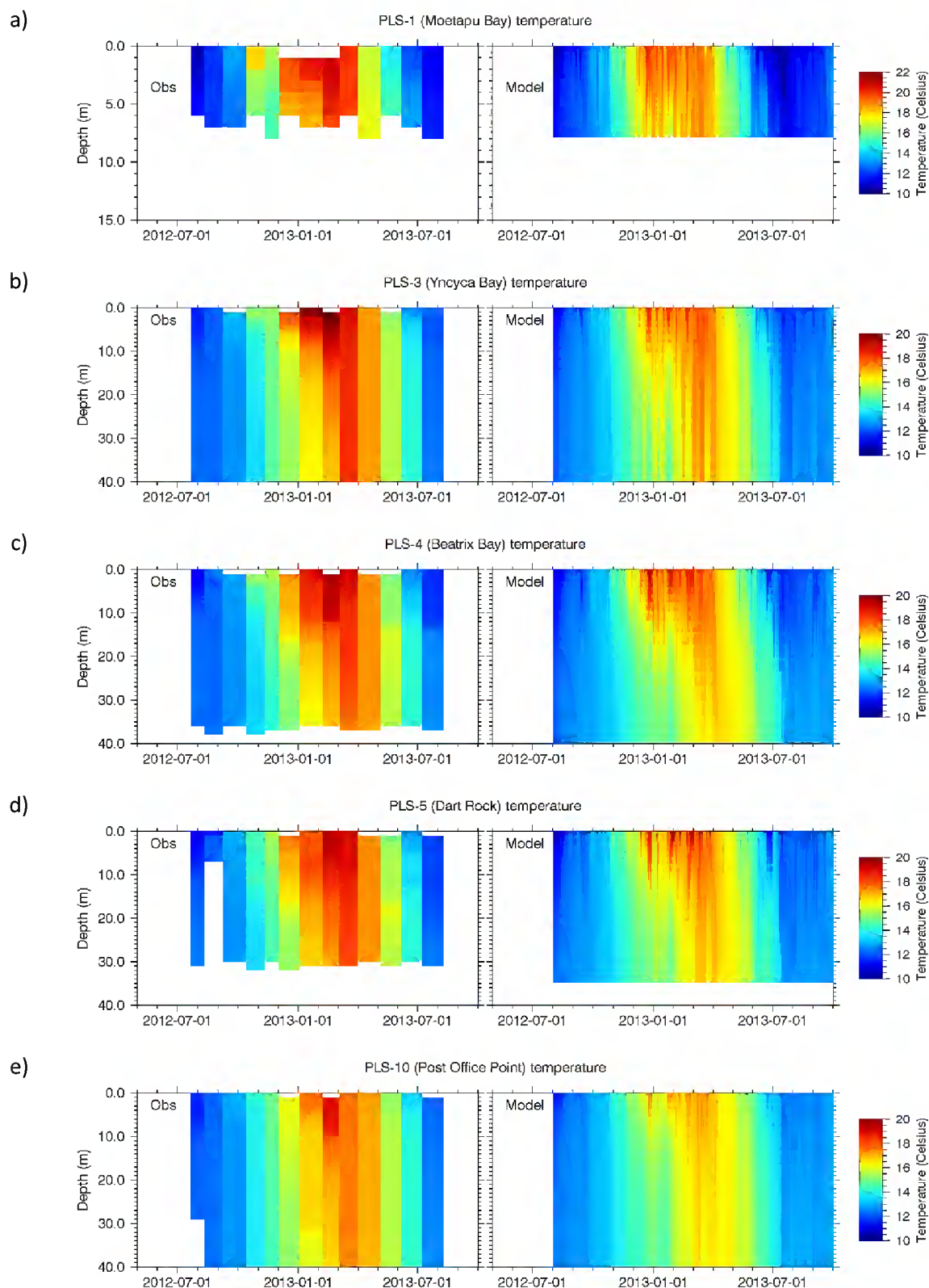


Figure 3-1: Observed and modelled temperature profiles. Temperature versus time and depth from monthly CTD casts (left) and model (right) for 5 sampling locations shown in Figure 2-7: a) PLS-1; b) PLS-3; c) PLS-4; d) PLS-5; e) PLS-10.

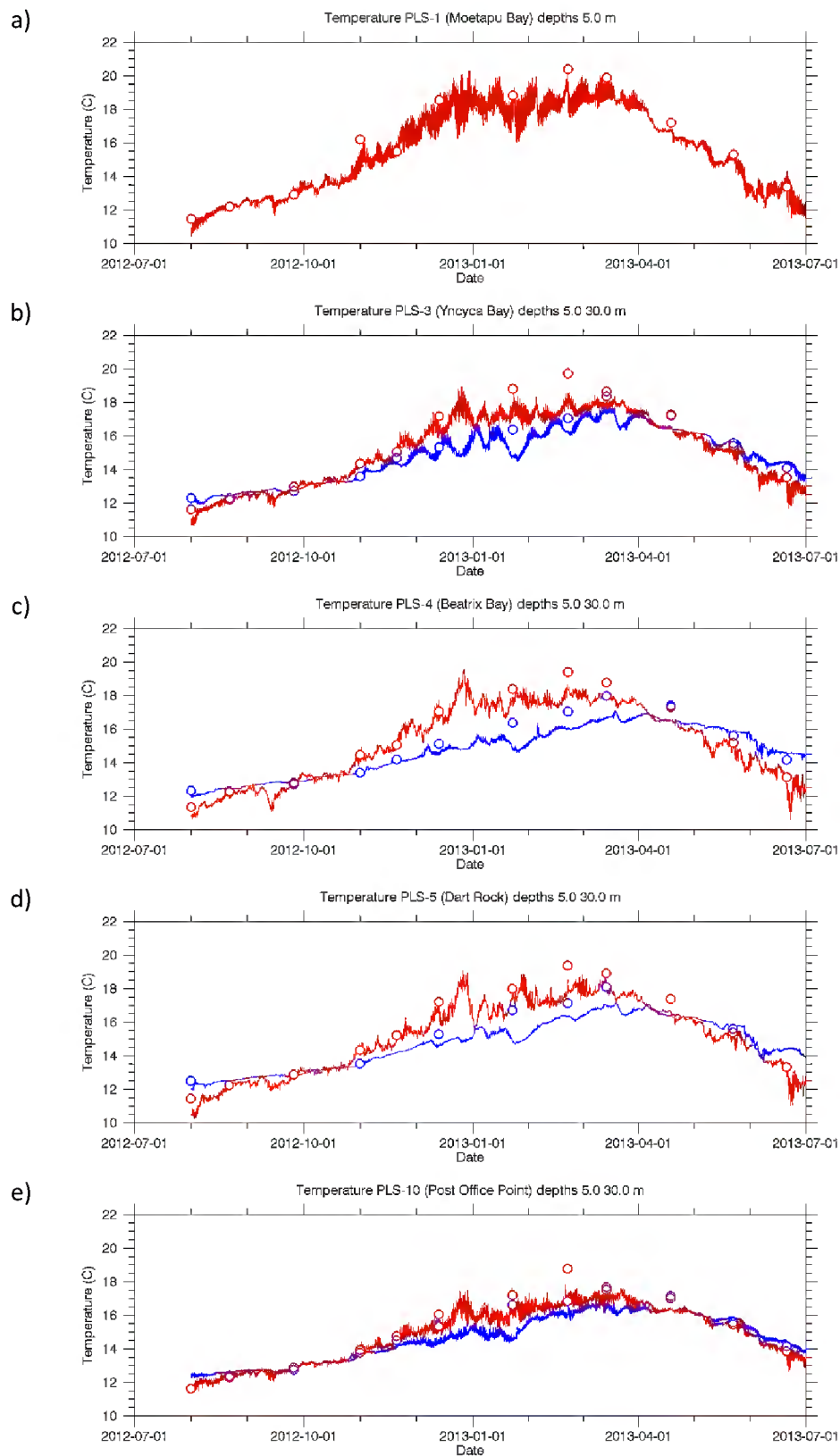


Figure 3-2: Observed and modelled temperature time series. Temperature at two depths (blue lower, red upper) from monthly CTD casts (symbols) and model (lines) for 5 sampling locations shown in Figure 2-7: a) PLS-1; b) PLS-3; c) PLS-4; d) PLS-5; e) PLS-10.

Figure 3-3 shows salinity¹⁴ versus time and depth at the same 5 CTD stations as Figure 3-1. All these plots show there is a low-salinity surface layer that is present much of the time, particularly in winter. Note that the salinity colour scale differs between the panels of Figure 3-3: at the innermost site (PLS-1 Moetapu Bay) the scale spans 20–35 g kg⁻¹ and at the outermost site (PLS-10 Post Office Point) it spans 30–35 g kg⁻¹. So the surface salinities fall as one moves from outer to inner Pelorus Sound, but the pattern of frequent surface freshening events occurs throughout the Sound. The timing of the surface freshening events agrees between the model and the measurements (bearing in mind the limitations of the monthly sampling) and the freshening events seem to follow pulses of Pelorus River flow (Figure 2-4).

Salinity time series at the mooring sites are shown in Figure 3-4. Agreement between model and measurements is very good. The model salinity is biased high at the outer site (PLS-10, Post Office Point) in the winter and spring of 2012. This might be a model spin-up issue: the model was initialised with zero freshwater on 24 May 2012 and it may take several months for the freshwater to spread through Pelorus Sound and into Cook Strait. The lack of freshwater input into the Cook Strait model might be an issue too.

A lack of freshwater in Pelorus Sound from late January to late April 2013 is apparent at all the sites. This coincided with a period of low Pelorus River flow (Figure 2-4) and a drought in central New Zealand¹⁵ and was terminated by a Pelorus River flood event with a peak flow of 500 m³ s⁻¹ on 22 April.

¹⁴ The term “salinity” in this report implies absolute salinity as defined by the TEOS-10 standard (Pawlowicz 2010).

¹⁵ <http://www.stuff.co.nz/business/farming/drought-recovery/8963794/Drought-worst-in-nearly-70-years>

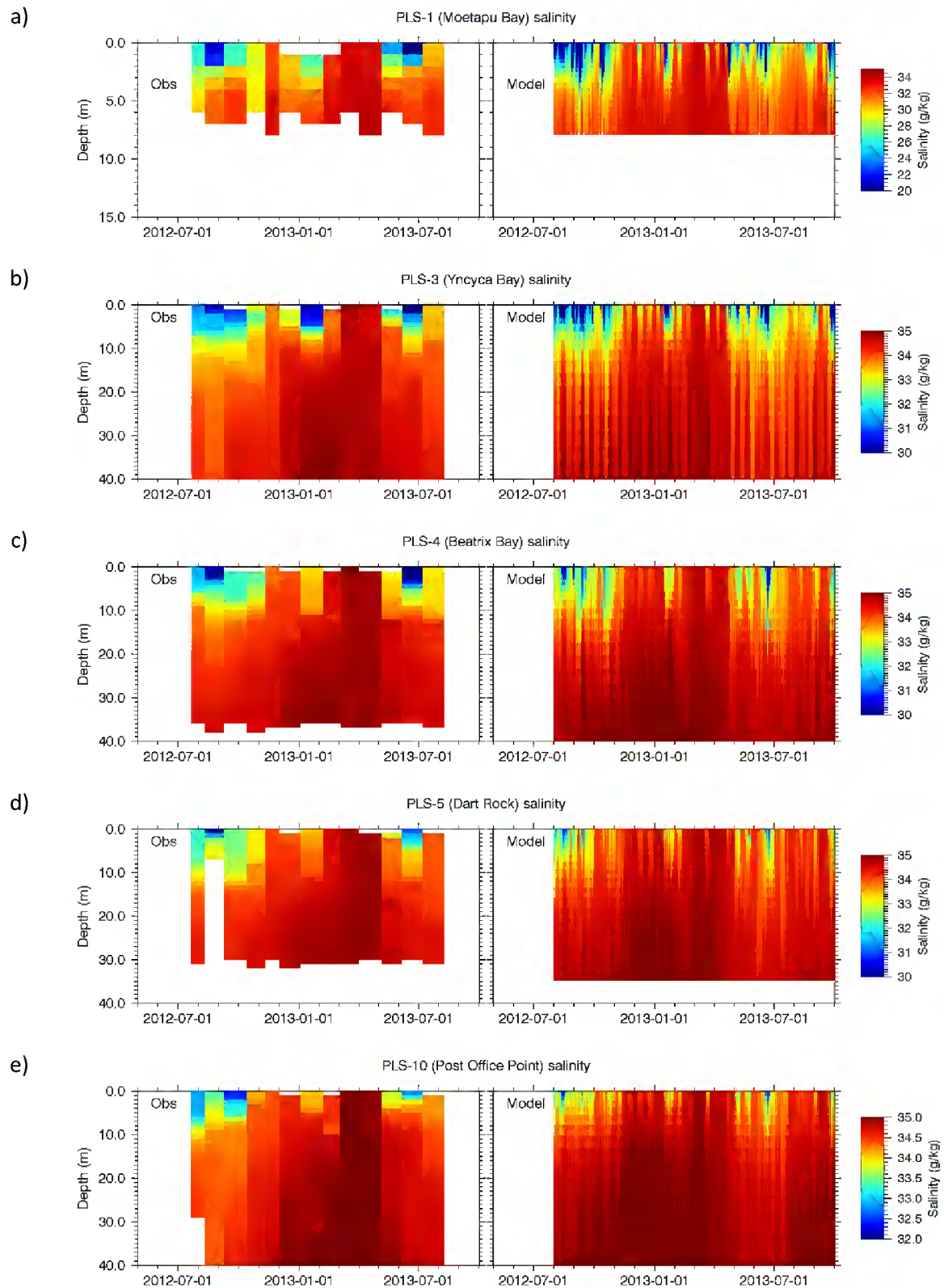


Figure 3-3: Observed and modelled salinity profiles. As Figure 3-1 but for salinity.

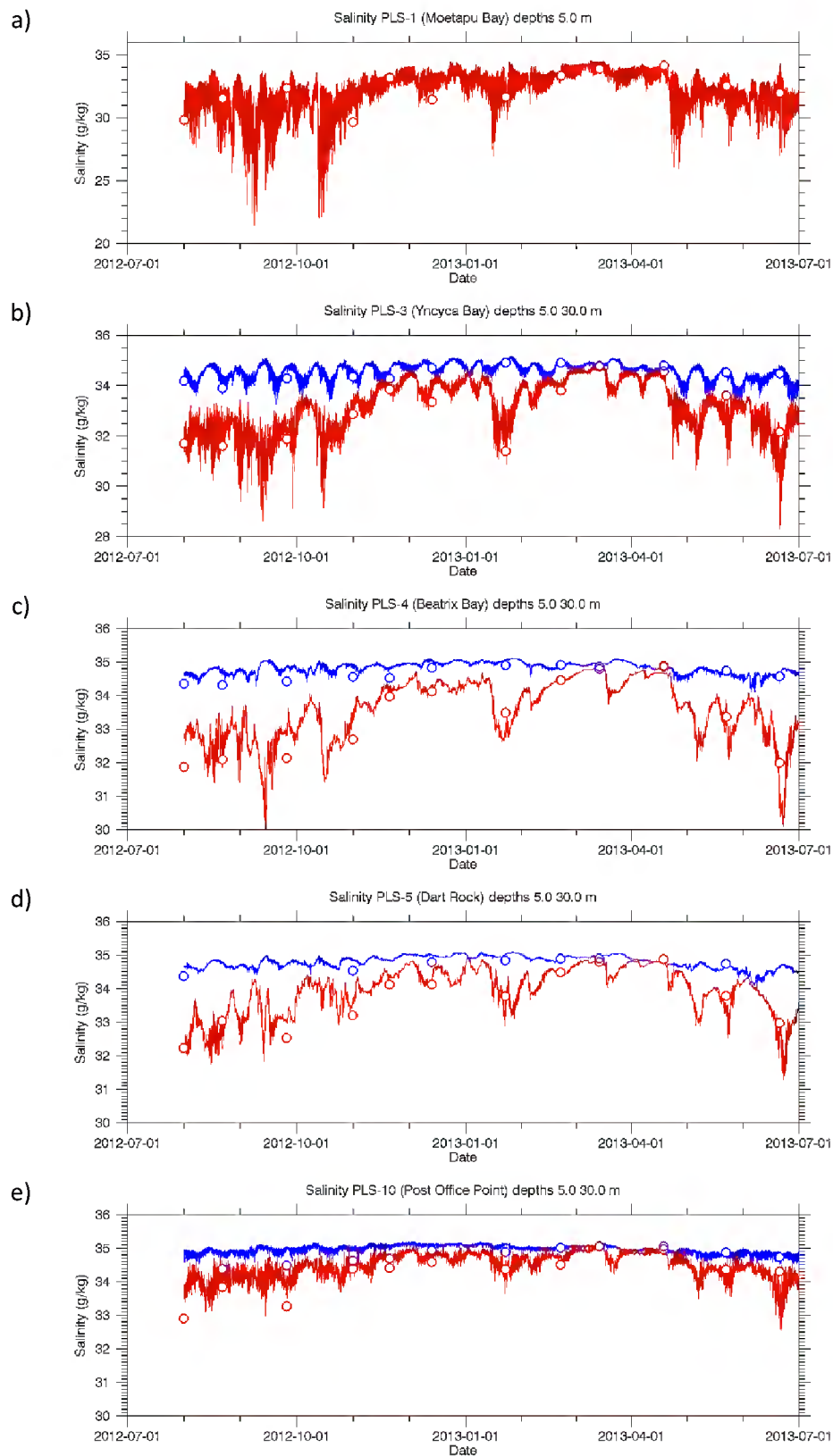


Figure 3-4: Observed and modelled salinity time series. As Figure 3-2 but for salinity.

3.2 Model vs observations: tidal height fluctuations

This section considers the accuracy of the model's representation of tidal fluctuations in sea surface height. These are estimated by fitting tidal harmonics of specified frequencies to the data. As is the case elsewhere around New Zealand, the dominant tidal constituent in the area is the lunar, semi-diurnal constituent (M2). The tidal variation is defined by two parameters: the amplitude (metres) and the phase (degrees) in time of the sinusoidal oscillation. A phase difference of 1° corresponds to a time difference of 1/360th of the tidal period: for the M2 tide, the period is 12.42 hours (0.5 lunar days) so a phase difference of 1° corresponds to a shift of 2.1 minutes.

3.2.1 Pelorus Sound 1994–1995 and 1997–1998 measurements

Table 3-1 compares measured and modelled M2 tidal parameters at tide gauge sites occupied in 1994–1995 and 1997–1998 (Figure 2-5). The model matches the amplitude within 6% and the phase within 1.5° (3 minutes) which represents very good agreement.

Table 3-1: Comparison of M2 tidal height parameters for Pelorus 1994–1995 and 1997–1998 tide gauges. M2 tidal sea level parameters from measurements and model. Here “ratio” means model value divided by observed value and “diffce” means model value minus observed value.

Tide gauge site and deployment	Record length (days)	Amplitude (m)			Phase (°)		
		Obs.	Model	Ratio	Obs.	Model	Diffce
Beatrix East 1994-1995 Deployment 55	54	0.878	0.927	1.06	270.2	271.6	1.3
Pelorus Entrance 1997-1998 Deployment 1	46	0.826	0.824	1.00	270.1	270.9	0.8
Pelorus Entrance 1997-1998 Deployment 2	69	0.868	0.867	1.00	271.7	272.1	0.4
Pelorus Entrance 1997-1998 Deployment 3	78	0.829	0.831	1.00	271.5	270.9	-0.6
Beatrix North 1997-1998 Deployment 1	46	0.881	0.886	1.01	270.6	271.7	1.1

Note that in Table 3-1, there are separate estimates of the M2 amplitude at one site (Pelorus Entrance) for each of the three 1997–1998 deployments and these estimates differ from each other, varying between 0.826 and 0.868 m. If one estimates the M2 constituent for a sufficiently long period (about one year), one gets a stable estimate representing the true value for that location, which can then be used for tidal predictions well into the future. (The situation is slightly complicated by the nodal variation, which is an astronomically-controlled oscillation of a few percent in amplitude and a few degrees in phase over an 18.6 year period.) However the purpose of the present calculations is not to estimate the true M2 tide, but to compare the model against observations. The M2 tide calculated over a period of a few months will vary from the true value, but ideally the model will capture that variation as long as the model and observations are analysed over the same period.

Table 3-2 presents a similar comparison for the S2 (solar, semi-diurnal) tidal constituent, which has a period of 12 hours and is the largest constituent after M2. Superposition, or “beating”, of the M2 and S2 constituents accounts for most of the spring-neap cycle in the semi-diurnal tide. Agreement is not quite as good as with the M2 constituent. The model tends to overestimate the amplitude slightly (implying that the spring-neap variation is being overestimated), with the largest discrepancy being

+13% at the Beatrix East gauge; the phase matches within 4°. It is normal for hydrodynamic models to agree less with the smaller constituents than with the M2, and this still represents good agreement.

Table 3-2: Comparison of S2 tidal height parameters for Pelorus 1994–1995 and 1997–1998 tide gauges. As Table 3-1 but for the S2 constituent.

Tide gauge site and deployment	Record length (days)	Amplitude (m)			Phase (°)		
		Obs.	Model	Ratio	Obs.	Model	Diffce
Beatrix East 1994-1995 Deployment 1	54	0.308	0.346	1.13	318.0	314.9	-3.1
Pelorus Entrance 1997-1998 Deployment 1	46	0.301	0.316	1.05	333.8	337.5	3.7
Pelorus Entrance 1997-1998 Deployment 2	69	0.362	0.387	1.07	310.5	309.5	-1.0
Pelorus Entrance 1997-1998 Deployment 3	78	0.233	0.230	0.99	323.0	323.5	0.5
Beatrix North 1997-1998 Deployment 1	46	0.323	0.342	1.06	334.6	338.4	3.8

Similar comparisons are presented in Appendix D for N2 (a semi-diurnal constituent) and O1 (typically the largest of the diurnal constituents) constituents. For N2 (Table D-1) the amplitude is within 8% and the phase is within 6°, which is very good agreement for one of the smaller semi-diurnal constituents. For O1 (Table D-2) the disagreement is larger (up to 46% in amplitude and 27° in phase) but given that the amplitude of this constituent is only 0.01–0.02 m, this level of disagreement is not unexpected and does not indicate a deficiency in the model's description of important hydrodynamic processes.

3.2.2 FRIA 2005 measurements

Table 3-3 compares measured and modelled M2 tidal parameters, the former calculated from ADCP pressure data collected during the FRIA project (Section 2.5.2, Figure 2-6). The FRIA ADCPs were deployed for short periods, the longest being 29 days (site 10 deployment 1); records shorter than 10 days were omitted from this analysis. This leads to quite a lot of variation in the amplitude estimated for the M2 tide, as explained above. Despite this variation, the model agrees with the observations reasonably well, tending however to overestimate the amplitude, by 7–12%. The phase agrees very well. Given the short record lengths, these comparisons should be given less weight than the longer-record comparisons in Section 3.2.1.

Table 3-3: Comparison of M2 tidal height parameters for FRIA 2005 ADCP pressure data. M2 tidal sea level parameters from measurements and model. Here “ratio” means model value divided by measured value and “diffce” means model value minus observed value.

ADCP site and deployment	Record length (days)	Amplitude (m)			Phase (°)		
		Meas.	Model	Ratio	Meas.	Model	Diffce
Site 1 deployment 2	10	0.684	0.731	1.07	276.5	276.0	-0.5
Site 11 deployment 2	11	0.742	0.801	1.08	276.8	276.8	0.0
Site 10 deployment 1	29	0.809	0.909	1.12	272.6	271.7	-0.9
Site 9 deployment 1	14	0.811	0.901	1.11	263.0	263.4	0.4
Site 8 deployment 2	10	0.675	0.748	1.11	276.7	276.6	-0.1
Site 6 deployment 1	14	0.809	0.897	1.11	96.5	95.8	-0.7
Site 15 deployment 1	14	0.816	0.902	1.10	97.0	96.7	-0.3

The S2 and other constituents have not been calculated from the FRIA ADCP data because of the short record lengths.

3.2.3 Havelock tide gauge data

Table 3-4 presents another comparison of measured and model tidal height parameters, this one using the Havelock tide gauge data (Section 2.5.3). The record length for the tide gauge analysis was 237 days (21 April to 14 December 2009) and for the model analysis it was 365 days (6 October 2012 to 6 October 2013). These record lengths are sufficient to permit stable estimates of the major constituents, therefore it is not necessary to match the modelled and measured time intervals as was done for the Pelorus Sound and FRIA data in the preceding sections.

Table 3-4: Comparison of tidal height parameters for Havelock tide gauge data. Tidal sea level parameters for 6 constituents from measurements and model. Here “ratio” means model value divided by measured value and “diffce” means model value minus observed value.

Constituent	Amplitude (m)			Phase (°)		
	Meas.	Model	Ratio	Meas.	Model	Diffce
M2	0.931	1.057	1.14	277.8	276.0	-1.7
S2	0.340	0.366	1.08	333.2	330.8	-2.4
N2	0.147	0.168	1.14	269.1	264.5	-4.6
K2	0.104	0.073	0.70	311.0	330.8	19.7
K1	0.051	0.051	1.00	340.0	345.7	5.8
O1	0.028	0.017	0.63	268.0	250.5	-17.5

For the M2 constituent the model matches the measured phase to within 2° (~4 minutes) but overestimates the amplitude by 14%. Similar performance is achieved for the two next-largest semi-diurnal constituents, S2 and N2. The model also performs quite well for the largest diurnal constituent, K1, but substantially underestimates the K2 and O1 constituents. Regarding the K2 constituent, it is very close in period to the S2 constituent: the beat period (the period over which the two constituents move in and out of phase with each other) is 182.5 days. This means that accurate estimates of the two constituents requires a long record, and the record lengths here are

marginal. For the O1 constituent, it is known that the NIWA EEZ tidal model that provided boundary forcing for this model does not reproduce this constituent particularly well in Cook Strait (Stanton, Goring, Bell 2001). Overall, the important result of this comparison is that the model somewhat overestimates the amplitude of the major semi-diurnal constituents.

3.3 Model vs observations: tidal velocity fluctuations

Tidal velocity variations are conventionally characterised by tidal ellipses, a representation indicating the path taken by the tip of a tidal current vector, which rotates at a constant angular frequency and changes in length (current speed) through a tidal cycle. A tidal ellipse is defined by four parameters:

- **Semi-major amplitude (m s^{-1}):** The semi-major axes are lines from the origin to the two most distant points on the ellipse perimeter. The two axes are equal in length, and this length represents the amplitude of the velocity along the semi-major direction.
- **Eccentricity:** At right angles to the semi-major axes are the semi-minor axes, which connect the origin to the two closest points on the ellipse perimeter. The eccentricity, or “fatness”, of the ellipse is the ratio of semi-minor to semi-major axis lengths. The eccentricity can be positive (vector rotates anti-clockwise) or negative (clockwise).
- **Inclination ($^{\circ}\text{T}$):** The inclination is the orientation of one of the semi-major axes. The choice between the two is arbitrary: here we take the semi-major axis directed towards the north-eastern or south-eastern quadrant and express the inclination as the orientation in degrees clockwise from true north ($^{\circ}\text{T}$).
- **Phase ($^{\circ}$):** The phase relates to the time at which the rotating tidal current vector passes through the semi-major axis. A phase difference of 1° corresponds to a time difference of $1/360^{\text{th}}$ of the tidal period.

The following subsections compare modelled tidal ellipses in Pelorus Sound with measurements from the same measurement campaigns as the section on tidal height fluctuations. Sample graphs are shown below, with a more complete set of graphs and tables in Appendix D.

3.3.1 Pelorus Sound 1994–1995 and 1997–1998 measurements

Suitable data for a tidal velocity comparison are available from three sites, all in the 1997–1998 campaign: Pelorus Entrance, Pelorus Tawero and Beatrix West (Figure 2-5). Of the full set of 3 sites \times 3 deployments \times 2 levels, 11 time series are suitable for the comparison. Figure 3-5 compares measured and modelled M2 tidal ellipses at the Pelorus Tawero site during deployment 1, when both the near-surface (9 m below the surface) and near-bottom (5 m above the bottom) meters returned good data. The measured and modelled ellipses match reasonably well in orientation and the model appears to under-predict the amplitude somewhat. At the near-surface meter the modelled ellipse is somewhat “fatter” (higher eccentricity) than the measured one. A tabular comparison (Appendix D, Table D-3) confirms these qualitative observations showing *inter alia* that the model over-predicts the amplitude by 30% at the surface and 20% at the bottom, that the model matches the measured inclination to within 3° and that the model phase leads the observations by 10° (20 minutes). This is a reasonable performance, given that the Pelorus Tawero site is at a location where the main channel of Pelorus Sound takes a sharp bend and tidal velocities can be expected to vary over small distances. An interesting feature of this location (model and measurements) is the large difference ($45\text{--}50^{\circ}$) in the inclination of the tidal ellipse between the near-surface and near-bottom meters.

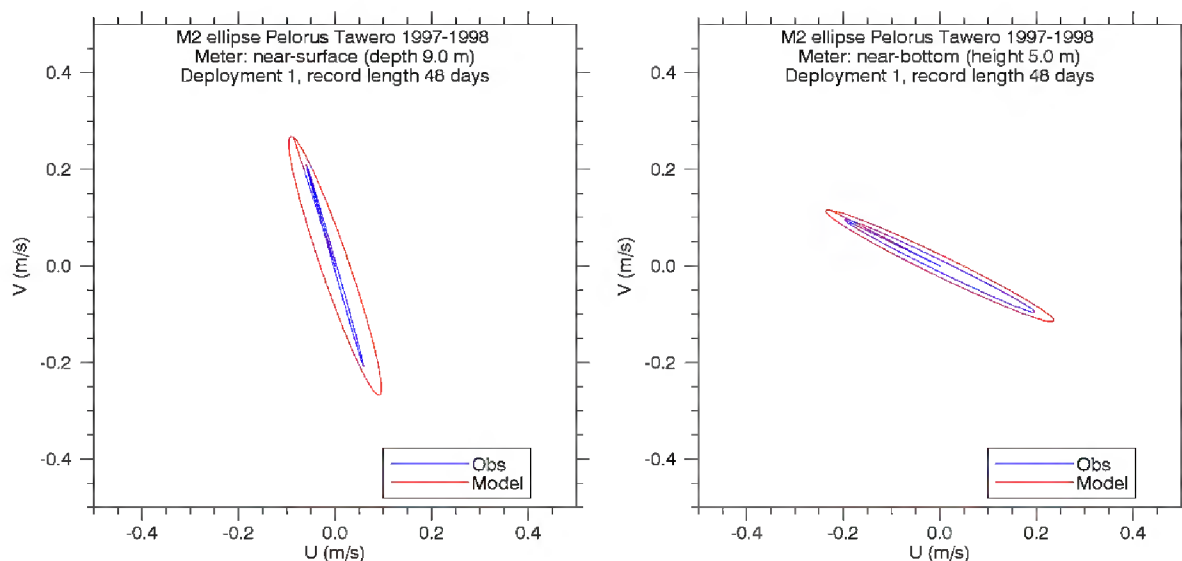


Figure 3-5: M2 tidal velocity comparison (Pelorus Tawero, deployment 1). M2 tidal ellipses from current meters (blue) and model (red). The axes correspond to the velocity components towards due east (u) and due north (v). The ellipses represent the magnitude and orientation of the tidal velocity variations (see text) and the straight line from the origin to the ellipse represents the phase. The left- and right-hand panels are for the near-surface and near-bottom current meters, respectively.

The remaining model-measurement comparisons are presented in Appendix D, Figure D-1 to Figure D-3 and Table D-3. At the Pelorus Entrance site there is a full set of 3 deployments \times 2 levels. The model under-predicts the amplitude by 10–30% and the inclination by 1–10%. At the lower meter the model eccentricity is biased high and the phase is biased high by 20° for deployments 1 and 2 (but only 5° for deployment 3). At this site the instrument metadata indicates a water depth of 68 m, but the model depth interpolated to the same location is 57 m, which suggests significant discrepancies in the bathymetry. (We don't know which, if either, is correct.) Specifically, the model grid (Figure 2-1) indicates that the site is on the northern flank of a bank at about 50 m depth, with a channel 600 m to the north at a depth of 80–90 m and the contours and spot depths on LINZ chart NZ6152 confirm this. In this situation a small error in the instrument position or a small error in the model's bathymetry will affect the tidal currents significantly.

At the Pelorus Tawero site we have already seen the deployment 1 data above; the addition of data from the lower meter in deployment 3 does not change the picture significantly. At Beatrix West there are near-surface datasets from two deployments. The model does reasonably well, over-predicting the amplitude by 10% and 33% and the phase by 13°, but getting the eccentricity and the inclination about right.

S2 current ellipses have been calculated and are presented in tabular form (only) in Table D-4. Overall results are similar to those for the M2 constituent (e.g., amplitude is under-predicted at Pelorus Entrance and over-predicted at Pelorus Tawero) but with somewhat more scatter.

3.3.2 FRIA 2005 measurements

An M2 tidal velocity comparison has been carried out for the 9 suitable time series from the FRIA 2005 campaign. These measurements were made with profiling instruments (ADCPs) rather than current meters. The quantity being compared is the vertical average between 85% and 20% of the water depth. (The former limit is intended to reduce the impact of discrepancies in water depth

between the instrument metadata and the model; the latter is intended to avoid contamination of the ADCP data by surface reflections.) The results are presented in Figure D-4, Figure D-5 and Table D-5. The comparison shows a level of agreement similar to that found in Section 3.3.1. The model/measurement amplitude ratio varies from 0.89 to 1.33, with more over-predictions than under-predictions. The inclination difference is between -3.8° and $+8.4^\circ$. The phase difference is between -9.0° and $+19.1^\circ$, with the highest value being an outlier.

3.4 Model vs observations: subtidal velocity fluctuations

This section considers the accuracy of the model's representation of sub-tidal currents, i.e., fluctuations in the currents with frequencies below the tidal frequency. Sub-tidal currents were estimated by taking the same velocity data that were tidally analysed in Section 3.2.3, but instead applying a low-pass temporal filter, an operation known as detiding. The filter was the 24G113 filter from Thompson (1983), applied to hourly values; see Figure 1 of that article for its frequency response. The filter removes essentially all fluctuations with a period of less than 2 days from the data and yields rather smooth time series as a result. Note that a comparison between model and measurements, as here, should not be particularly sensitive to the filter characteristics as long as the same filter is applied to both.

3.4.1 Pelorus Sound 1994–1995 and 1997–1998 measurements

Figure 3-6 compares measured and modelled scatter plots of the sub-tidal velocities at the Pelorus Tawero upper current meter during deployment 1. (A tidal vector analysis of the same time series is shown above in Figure 3-5, left-hand panel.) The red ellipse in each scatter plot is a variance ellipse, a conventional representation of the magnitude and pattern of variability in velocity data. A variance ellipse can be characterised by its semi-major axis (in this context called a principal axis), eccentricity and inclination, like a tidal ellipse. However a variance ellipse does not have a phase (since it says nothing about the timing of the variability) and its eccentricity has no sign (since it says nothing about the rotation of velocity vectors). Also, the centres of the variance ellipses are offset from the origin by an amount representing the mean current over the period of the deployment.

Figure 3-6 indicates that both the measured data and the model have a mean of 0.08 m s^{-1} towards a direction of N to NNW. (The numeric values for the mean flow vector are listed in Table D-6.) The variance ellipses are of similar size—indicating broadly similar magnitudes of variability—but different shape, the measured one being elongated to NW-SE (probably because of a large excursion in the data points to the SE) and the modelled one being nearly circular.

To illustrate how the measured and modelled time series match up in time, a direction was first specified (subjectively) for each current meter site representing local channel direction. The directions were: 60° T for Pelorus Entrance; 320° T for Pelorus Tawero; and 50° T for Beatrix West. Figure 3-7 shows separate time series plots of the velocity components along the channel (towards 320° T) and across the channel (towards 230° T). Generally the model produces comparable fluctuations to the measured data in the along-channel and across-channel directions, with several peaks coinciding between model and data, but several not coinciding. Conspicuously, the large negative peak in the along-channel current meter time series (which appears to be real and is probably caused by a wind event) is not reproduced by the model.

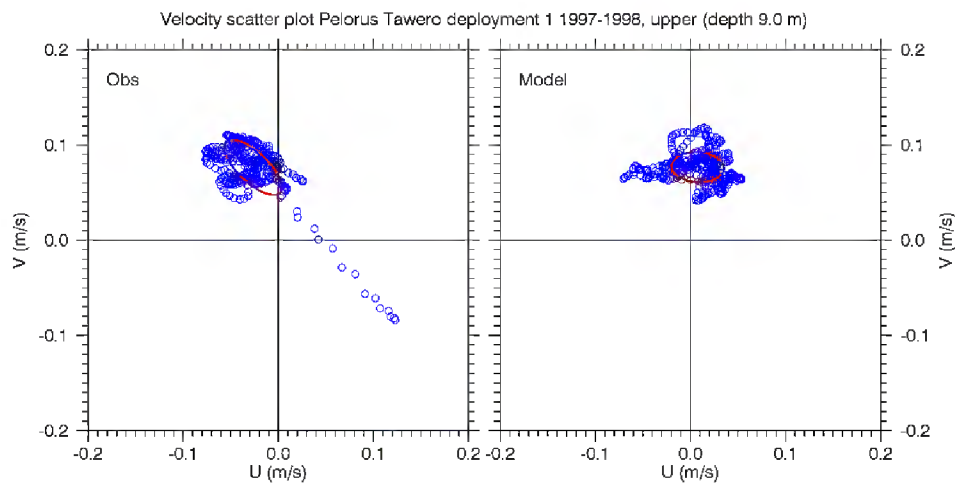


Figure 3-6: Sub-tidal velocity vector comparison (Pelorus Tawero deployment 1). Scatter plots of measured (left) and modelled (right) sub-tidal velocity at the Pelorus Tawero upper current meter during deployment 1. The axes correspond to the velocity components towards due east (u) and due north (v). The red lines are variance ellipses, representing the magnitude and orientation of the sub-tidal velocity variations (see text).

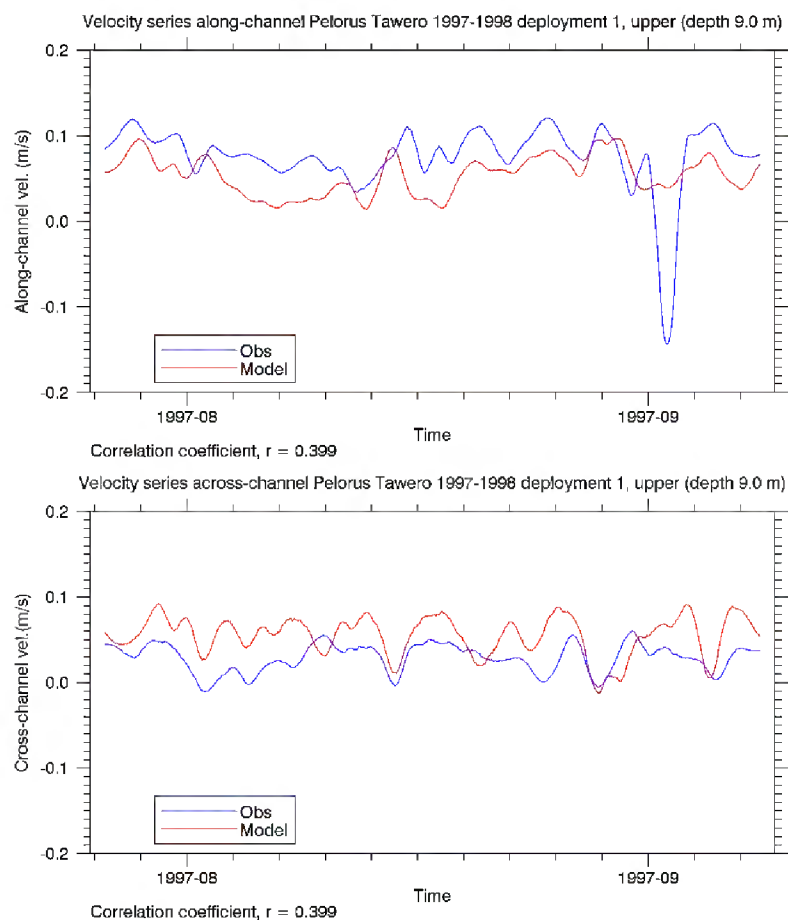


Figure 3-7: Sub-tidal along-channel and across-channel velocity comparisons (Pelorus Tawero deployment 1). Time series of measured (blue) and modelled (red) sub-tidal velocity components in the along-channel (towards 320° T, upper panel) and across-channel (towards 230° T, lower panel) directions at the Pelorus Tawero upper current meter during deployment 1.

The degree of closeness of the match between modelled and measured fluctuations is quantified on the graph with the temporal correlation coefficient, r , which is 0.399 for both directions. (The exact agreement here is a coincidence.) An r value of 0.399 implies an r^2 of 0.16, i.e., the model explains 16% of the variance in the measured data. This is a modest level of agreement, and may well arise by chance.

Appendix D includes a full set of scatter-plot comparisons (Figure D-6 to Figure D-9) along with tabulated parameters in Table D-6. The model does a reasonably good job of reproducing the observed estuarine circulation at Pelorus Entrance (upper flow to NE, lower flow to SW) and Pelorus Tawero (upper flow to N, lower flow to S or SE), although it consistently underestimates the magnitude of the lower, SW flow at Pelorus Entrance. At Beatrix West the mean flow is relatively weak in both model and measurements. Magnitudes of variability are broadly similar between model and measurements. Temporal correlations (Table D-6) are variable, but occasionally large, e.g., 0.8 at Pelorus Entrance deployment 3, lower meter.

3.5 Currents and volume fluxes

The capacity of the environment to dilute and disperse additional dissolved material—whether it be fish farm wastes or substances from other sources—is clearly central to the present project. Before moving to the biophysical model and its results, the remainder of this section presents some relevant analyses of the currents in the main channels of the Pelorus Sound system. A later section (Section 3.6) looks specifically at the transport of dissolved material through and out of the Sound.

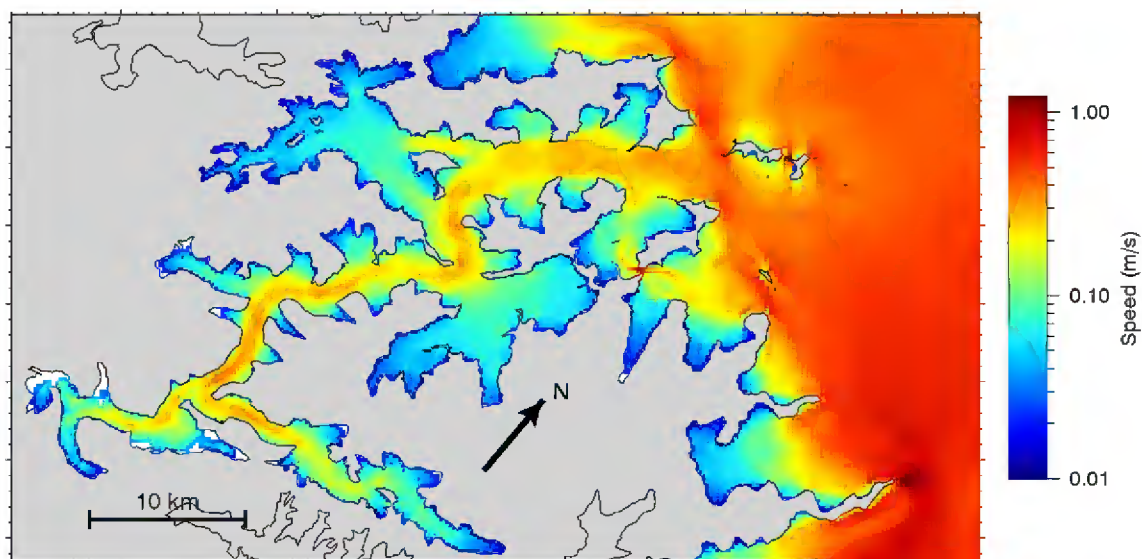


Figure 3-8: Model mean current speed. Mean current speed at 5 m depth, based on one year's hourly data from the 200 m model.

As an approximate indicator of near-field dispersal of nutrients or waste from a mussel farm or fish farm, Figure 3-8 shows the mean current speed at 5 m depth. The largest mean speeds ($\sim 1 \text{ m s}^{-1}$) are associated with the strong tidal currents in Cook Strait. There is a band of currents around $0.2\text{--}0.3 \text{ m s}^{-1}$ through the main channel of Pelorus Sound and into Kenepuru Sound. This band is produced largely by the tides, but the estuarine flow of surface water out of the Sound also contributes.

The analyses in the remainder of this section deal with several sections across Pelorus Sound (Figure 3-9).

Figure 3-10 shows the volume flux through the section labelled Boat Rock Point in Waitata Reach, based on hourly model output. The flux through the Pelorus Sound entrance (Boat Rock Point) is typically $50\text{--}60,000\text{ m}^3\text{ s}^{-1}$ at spring tide and $20\text{--}30,000\text{ m}^3\text{ s}^{-1}$ at neap tide. Given that the volume of Pelorus Sound (in the region defined for the flushing calculations of Section 3) is around $10,300 \times 10^6\text{ m}^3$, the peak spring-tide volume flux through Waitata Reach would be large enough to replace all the water in the Sound in 2.0 days, if it were maintained for long enough. Of course, the peak tidal transports are not maintained for several days and fluctuating tidal fluxes are not efficient at flushing the Sound. The true flushing time for Pelorus Sound is on the order of 30–50 days (see Section 3.6 and Table 3-5).

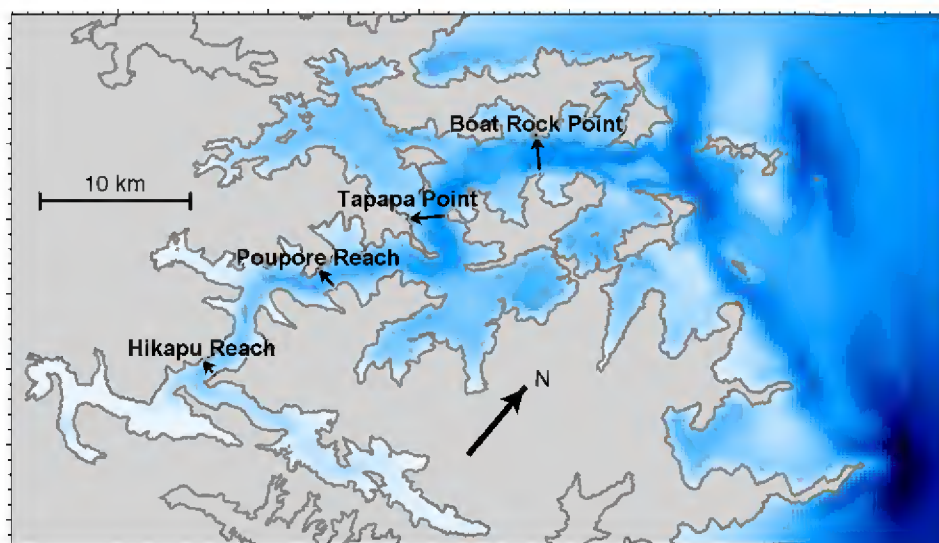


Figure 3-9: Location of sections used for velocity and volume-flux analyses. A map showing the model bathymetry and land mask (200 m grid), with labelled cross-sections.

The black lines in Figure 3-10 are based on a moving-window analysis for the semi-diurnal tide, with the central black line indicating the sub-tidal part of the volume flux. Because Pelorus Sound inside the Boat Rock Point section is a single bay with no other entrance, the sub-tidal flux is required by volume conservation considerations to be very small.

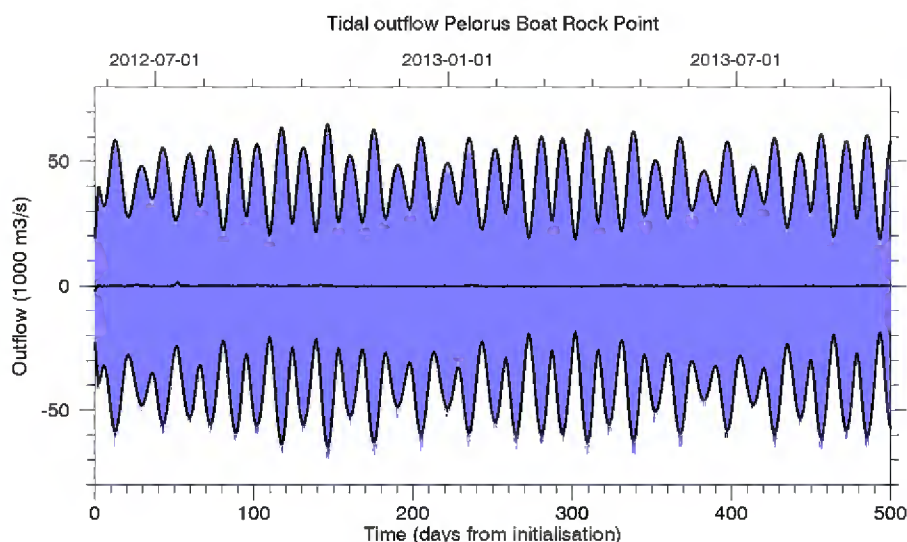


Figure 3-10: Tidal volume fluxes. Time series of volume flux for sections through the Boat Rock Point section across Waitata Reach. The light blue line represents the hourly volume flux (outflow positive) and the thick black lines represent the mean, plus & minus the amplitude of the semi-diurnal tidal flux as estimated by a moving-window tidal harmonic analysis (window width 3.5 days).

The information on volume fluxes presented in this section relates to the vertically-averaged currents. Another important aspect of the currents in Pelorus Sound is the vertical variation. Figure 3-11 shows plots of time-averaged velocity (positive outwards) on four sections across the main channel (see Figure 3-9 for the locations). In all cases there is a layer of outwards flow ($\sim 0.2 \text{ m s}^{-1}$) overlying a layer of inwards flow ($\sim 0.1\text{--}0.2 \text{ m s}^{-1}$). This is the vertical structure expected for an estuarine circulation, in which outward-moving brackish water overlies inward-moving saline water. At the outermost section across Waitata Reach (Boat Rock Point, Figure 3-11d), the interface between the inward and outward flows (the zero-velocity level) is 20 m below the surface and nearly horizontal. At the three sections further inside Pelorus Sound the outflow is displaced to one side or other of the section as the outflowing water snakes through the Sound. Animations (not shown) of these graphs with monthly average data indicate that this vertical structure is set up within 30 days or so of the beginning of the simulation and continues with relatively little change throughout.

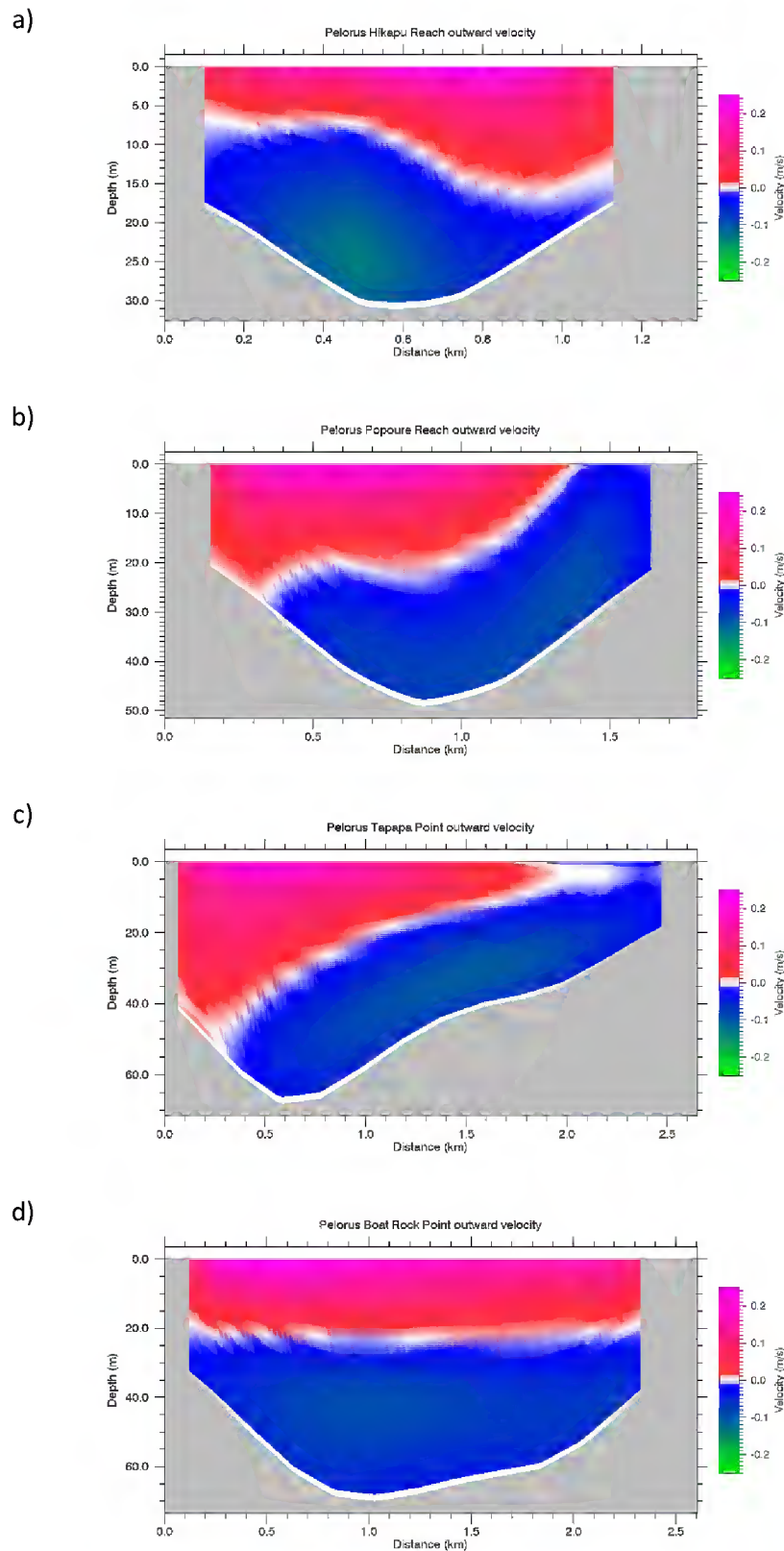


Figure 3-11: Velocity through cross-channel sections. One-year mean modelled velocity perpendicular to sections across Pelorus Sound as shown in Figure 3-9: a) Hikapu Reach; b) Popoure Reach; c) Tapapa Point (Tawhitinui Reach); d) Boat Rock Point (Waitata Reach). Velocity is positive outwards and the view is from the seaward side of the section, looking inwards.

The outwards velocity averaged over the top 20 m of the water column in the Boat Rock Point section is approximately 0.12 m s^{-1} and the width of the channel at this point is 2.2 km. This implies an outward flow in the estuarine circulation of $5300 \text{ m}^3 \text{ s}^{-1}$, which is an order of magnitude less than the volume flux in the peak spring tidal flow. However the estuarine circulation is very persistent, whereas the tidal flow reverses regularly, so we can expect the estuarine circulation to have a large effect on the flushing of tracers from Pelorus Sound.

3.6 Flushing

A set of simulations was set up to investigate the dilutive capacity of Pelorus Sound for idealised sources of dissolved material. Passive tracers, or virtual dyes, were injected into a hydrodynamic model of the Sound at five sites (Figure 3-12) distributed through Pelorus Sound. There was a release 5 m below the surface at all sites and at the three outer sites (Popoure Reach, Beatrix Bay and Waitata Reach) there was a second release 5 m above the bottom, giving a total of 8 virtual dyes. The model was run at two resolutions, 400 m and 200 m, for the same 500-day period in 2012–2013 as the biogeochemical simulations.

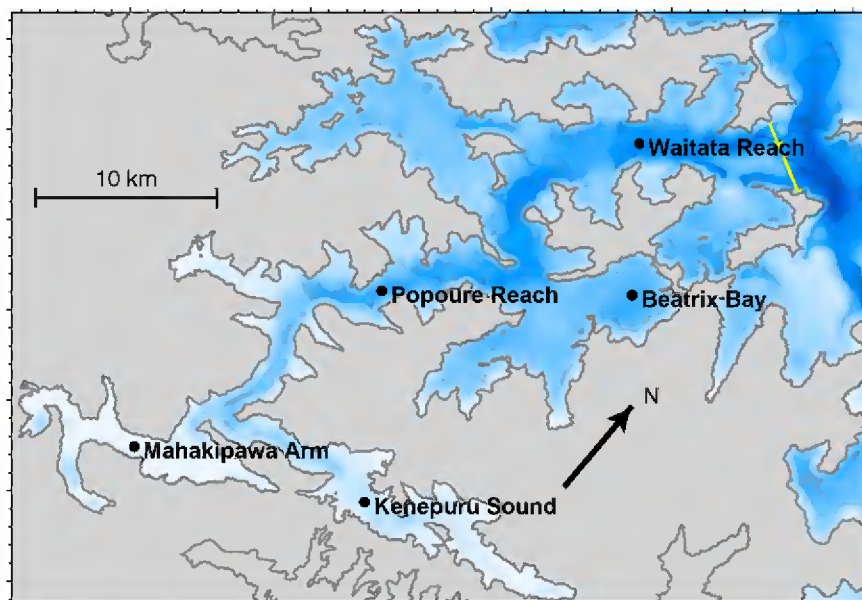


Figure 3-12: Location of passive tracer sources in the flushing simulations. A map showing the model bathymetry and land mask (200 m grid), with source locations for the flushing simulations (black circles and labels) and the boundaries for calculation of volume integrals (yellow lines across Pelorus Sound Entrance and Allen Strait).

The release rate Q of each dye was constant at a nominal 1 kg s^{-1} . The concentration C of the same dye at any location and time is measured in kg m^{-3} and should be proportional to Q (i.e., doubling the release rate should exactly double the concentration). Therefore the ratio between them, C/Q , can be called a normalised concentration: it has units of s m^{-3} and depends on the flow and the location of the source, but not on the release rate. It is convenient to represent this normalised concentration in terms of its reciprocal, called the dilution rate D , which has units of $\text{m}^3 \text{ s}^{-1}$ (Equation 3-1).

Equation 3-1: Definition of the instantaneous dilution rate

$$D = Q/C$$

A simple physical example illustrates the significance of the dilution rate. Consider a source of passive tracer, or dye, in a river. The dye plume will initially be narrow, but within a few hundred metres downstream (or kilometres for a large river, and assuming no major tributaries join in the meantime) the dye will become uniformly mixed across the river, with a concentration equal to the release rate divided by the river's flow rate. In other words, the dilution rate at large distances downstream is equal to the river flow rate. Within the dye plume closer to the source, the dilution rate is lower (the normalised concentration is higher), because not all of the river flow has mixed into the plume. Note that for a medium-sized river like the Pelorus the mean flow rate is approximately $50 \text{ m}^3 \text{ s}^{-1}$ and for the Clutha River, the largest river by volume in New Zealand, it is approximately $500 \text{ m}^3 \text{ s}^{-1}$.

Within the context of coastal inlets, it is common to introduce the concept of flushing time (Zimmerman 1988; Monsen, Cloern et al. 2002). Here we specify the boundaries of the inlet (as in Figure 3-12), calculate the volume V (in m^3) and evaluate the mass (in kg) of the tracer inside this volume. If the release rate is kept steady for long enough, we expect the mass to reach a more or less steady equilibrium value M_e . The equilibrium flushing time T_e is then defined by Equation 3-2:

Equation 3-2: Definition of flushing time.

$$T_e = M_e / Q$$

This gives a result in seconds, which is normally converted to days for convenience.

From the equilibrium mass M_e and the volume V we can calculate the equilibrium mean concentration $C_e = M_e / V$ and from that we can calculate an equilibrium dilution rate D_e (Equation 3-3), which is representative of the inlet as a whole and applies when there is a balance, more or less, between input of the tracer from the source and flushing through the boundaries.

Equation 3-3: Definition of the equilibrium dilution rate

$$D_e = \frac{Q}{C_e} = \frac{V}{T_e}$$

Although discussions of dilution in coastal inlets often concentrate on the flushing time, the equilibrium dilution rate is often a more pertinent measure, and it involves the inlet volume as well as the flushing time.

The concept of a flushing time originally came from consideration of a well-mixed water body, in which "clean" water from outside enters and is immediately mixed throughout, with the inflow balanced by an equal outflow of mixed water. On a laboratory scale, such a system is called an exponential dilution flask (Ritter and Adams 1976). If a tracer is initially mixed through such a water body, then its concentration will fall away with time according to an exponential curve:

Equation 3-4: Exponential dilution in a well-mixed water body

$$C(t) = C(0)e^{-t/T_e}$$

Thus after time T_e the concentration will have dropped to $1/e$ (0.37) times its initial value and after time $2T_e$ it will have dropped to $1/e^2$ (0.14) times its initial value. However coastal inlets are not well mixed (as is quite clear from the figures below) and so do not follow this equation exactly. In other words, we cannot say from the flushing time alone how quickly a dissolved substance will be eliminated from an inlet. Nevertheless the flushing time is still a reasonably good guide to the speed

with which dissolved material is flushed from the system: after one flushing time *most but not all* of the material initially in an inlet will have been flushed out; and complete flushing will generally take several flushing times.

For an indication of how these concepts can be applied to Pelorus Sound, Figure 3-13 shows the time series of the total tracer mass within the Sound for the 8 tracers in the 200 m simulation. The lower horizontal axis in the plot shows time in days from the beginning of the simulation (and the tracer release); the upper horizontal axis shows the date. As explained in connection with Figure 3-12, there are 5 release sites, with near-surface sources only at two sites and both near-surface and near-bottom sources at the remaining three. The vertical axis is the normalised mass of tracer within the Sound, i.e., it is the mass M (in kg) divided by the release rate Q (in kg s^{-1}), yielding a value in seconds, which is converted to days for plotting.

If Pelorus Sound were a simple, well-mixed volume, then all eight lines in Figure 3-13 would follow the same path. They would have an initial linear portion with a slope of 1 (i.e., one day's accumulation of mass per one day of release) and would then tend exponentially towards a horizontal line, at a value equal to the flushing time of the inlet. The actual lines do exhibit some of this behaviour, but deviate in several important respects. For the (black) line representing the tracer released in Mahakipawa Arm, near Havelock, the line rises with a slope of 1 for about 30 days (i.e., no tracer leaves the Sound in this time) and then flattens out at a value of around 35 days. There is a peak with a value of around 50 days in late April 2014. The accumulation of tracers up to that peak coincides with a period of low Pelorus River flow and low freshwater in Pelorus Sound, noted in Section 3.1.1.

The tracer released in Kenepuru Sound (grey) behaves similarly to the Mahakipawa Arm tracer with somewhat larger values. The Popoure Reach (blue) and Beatrix Bay (red) tracers also follow a similar pattern; it is apparent for these release locations that the near-surface (thick line) tracers have lower flushing times than the near-bottom (thin line) tracers. Finally, for the tracer released in Waitata Reach (dark green), the flushing time for the near-surface tracer release is very low, at around 10 days, whereas the tracer released near the bottom has an flushing time comparable to the others at 25–30 days.

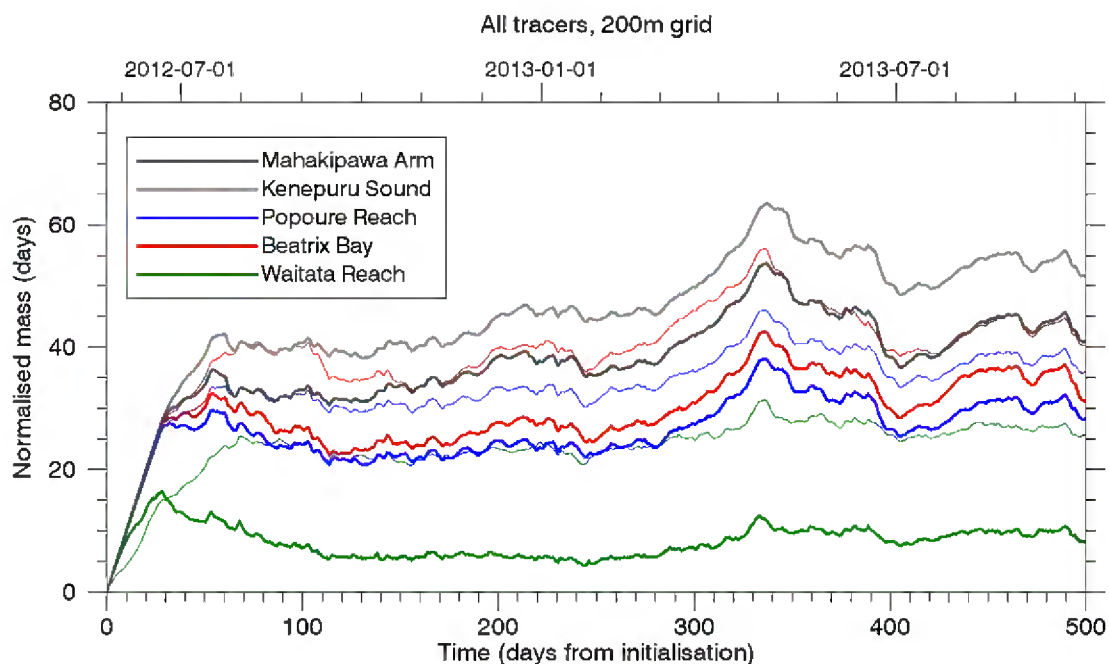


Figure 3-13: Accumulation of tracer from the 200 m flushing simulation. Normalised mass of tracer within Pelorus Sound versus time in the 200 m flushing simulation. See the figure legend for the relationship between the release location (Figure 3-12) and the line colour. For the three outer sites, the near-surface tracer is indicated by a thick line and the near-bottom tracer by a thin line of the same colour.

To estimate an annual-average flushing time and dilution rate for the tracers in Pelorus Sound, we have taken the average normalised mass for each tracer over the final 365 days of the simulation, i.e., from 16 October 2012 to 16 October 2013. The annual-average flushing time (Table 3-5) for the Mahakipawa Arm tracer is 41 days and for the Kenepuru Sound tracer it is 50 days. For the remaining tracers the flushing time is between 25 and 40 days, with the exception of the Waitata Reach near-surface tracer, which has a flushing time of only 7.8 days.

Table 3-5: Equilibrium flushing times and dilution rates for Pelorus Sound. Flushing times and dilution rates evaluated from the data in Figure 3-13 averaged over the last 365 days of the 200 m flushing simulation.

Site	Volume (10 ⁶ m ³)	Flushing time T_e (days)	Dilution rate D_e (m ³ s ⁻¹)
Mahakipawa Arm	10338	40.9	2930
Kenepuru Sound		49.9	2400
Popoure Reach near-surface		27.5	4350
Popoure Reach near-bottom		35.8	3340
Beatrix Bay near-surface		31.2	3840
Beatrix Bay near-bottom		42.0	2850
Waitata Reach near-surface		7.8	15,370
Waitata Reach near-bottom		25.2	4750

Previous estimates of the Pelorus Sound flushing time were discussed by Broekhuizen and Hadfield (2012) with reference to the evidence of Mr Ben Knight to the EPA King Salmon Hearing (Knight 2012b)—see Table 2-1 of Broekhuizen and Hadfield (2012). Discounting an outlier estimate of 12.9

days, based on questionable tidal prism arguments, there are several estimates between 21 and 49 days, which agree with the current model-based estimates, given the large uncertainties associated with the definition and estimation of flushing time.

It is also useful to compare the flushing behaviour between the 200 m model (which is used for production simulations in this project) and the 400 m model (which is used for development). This comparison is shown for the near-surface sources in Figure 3-14. The differences are very small. We have not run a 100 m model for this comparison (as was done for Queen Charlotte Sound) because the 100 m model is very expensive to run and Pelorus Sound does not have a region like Tory Channel, which has a major influence on mixing and clearly requires fine spatial resolution to be represented accurately in the model. In other words, we do not expect tracer dispersion in Pelorus Sound to be as sensitive to model resolution as it is in Queen Charlotte Sound, and Figure 3-14 supports that expectation.

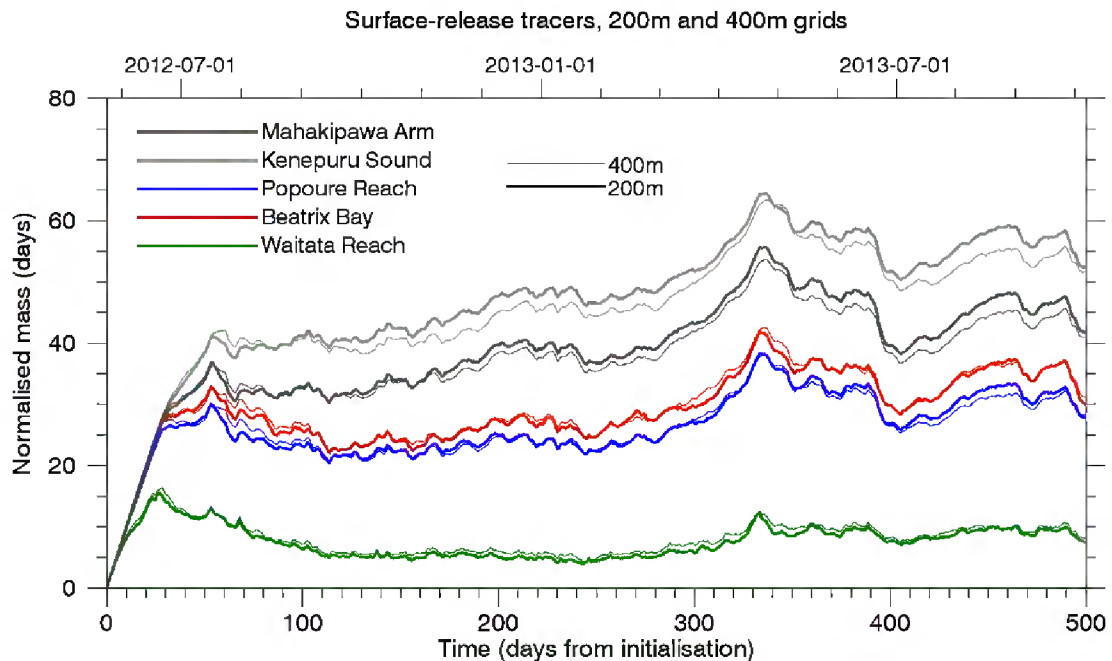


Figure 3-14: Effect of model resolution on flushing. Normalised mass of tracer within Pelorus Sound versus time for near-surface releases from the five sites, from the 400 m (thin) and 200 m (thick) models.

The next five figures (Figure 3-15 to Figure 3-19) show the mean surface concentration for each tracer, normalised and expressed as a dilution rate as described above. For the Mahakipawa Arm source (Figure 3-15) the dilution rate is $\sim 300 \text{ m}^3 \text{ s}^{-1}$ (dark red) and increases through $\sim 1000 \text{ m}^3 \text{ s}^{-1}$ (red–magenta) in Popoure Reach, $\sim 2000 \text{ m}^3 \text{ s}^{-1}$ (blue) in Tawhitinui Reach and $\sim 4000 \text{ m}^3 \text{ s}^{-1}$ (dark green) in Waitata Reach. Outside Pelorus Sound, the tracer plume bends north-westwards and leaves via the northwest boundary of the domain. For a source in Kenepuru Sound (Figure 3-16) the lowest dilution rates occur in Kenepuru Sound and not Mahakipawa Arm, but the pattern is otherwise identical.

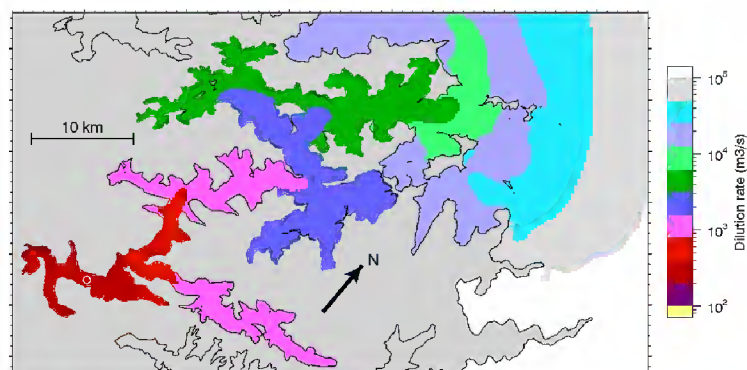


Figure 3-15: Equilibrium concentration for Mahakipawa Arm tracer. Surface concentration of tracer from the Mahakipawa Arm (near-surface) source in the 200 m model, averaged over the final 365 days and expressed as a dilution rate. The source location is indicated by a white circle.

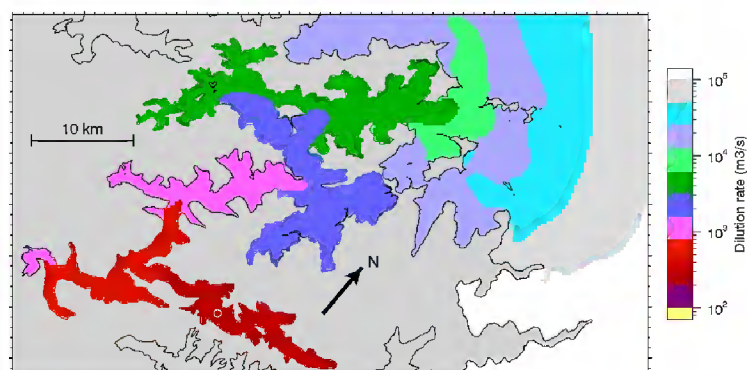


Figure 3-16: Equilibrium concentration for Kenepuru Sound tracer. As Figure 3-15 but for the tracer released in Kenepuru Sound.

For the Popoure Reach tracers the near-surface source (Figure 3-17a) produces a plume with dilution rate $\sim 1000 \text{ m}^3 \text{ s}^{-1}$ (magenta) extending seaward, whereas the near-bottom source (Figure 3-17b) produces a much more extensive area with similar dilution rates throughout Popoure Reach and inner Pelorus Sound. This is a result of the pronounced estuarine circulation in Pelorus Sound, taking surface water towards the sea, to be replaced by inward moving bottom water. In outer Pelorus Sound and Cook Strait the dilution rate pattern is indistinguishable for the near-surface and near-bottom sources, as for the Mahakipawa Arm and Kenepuru Sound tracers. In other words, however tracer material is injected into the inner part of Pelorus Sound, it ultimately moves out through the surface waters of outer Pelorus Sound in the same way.

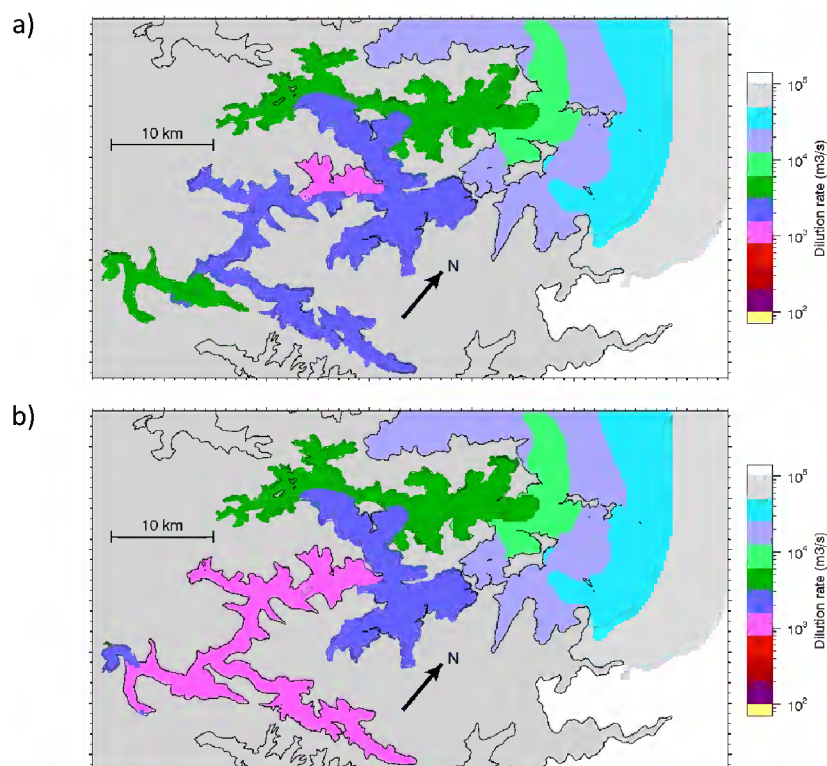


Figure 3-17: Equilibrium concentration for Popoure Reach tracers. Surface concentration of tracers from Popoure Reach (a) near-surface and (b) near-bottom tracer sources in the 200 m model, averaged over the final 365 days and expressed as a dilution rate. The source location is indicated by a white circle.

The Beatrix Bay tracers (Figure 3-18) produce similar patterns to the preceding ones but with the highest concentrations (lowest dilution rates) in Beatrix Bay.

For the tracers released in Waitata Reach, there is a very marked difference between the near-surface and near-bottom sources. The near-surface source (Figure 3-19a) produces a plume ($\sim 4000 \text{ m}^3 \text{ s}^{-1}$, dark green) that extends towards Cook Strait and also into Port Ligar, with relatively low concentrations (high dilution rates) everywhere else. The near-bottom source (Figure 3-19b) fills Pelorus Sound with tracer at $\sim 4000 \text{ m}^3 \text{ s}^{-1}$. As with the Popoure Reach tracers, this indicates the effect of the estuarine circulation, which takes near-surface water out of the Sound, but takes near-bottom water into the Sound.

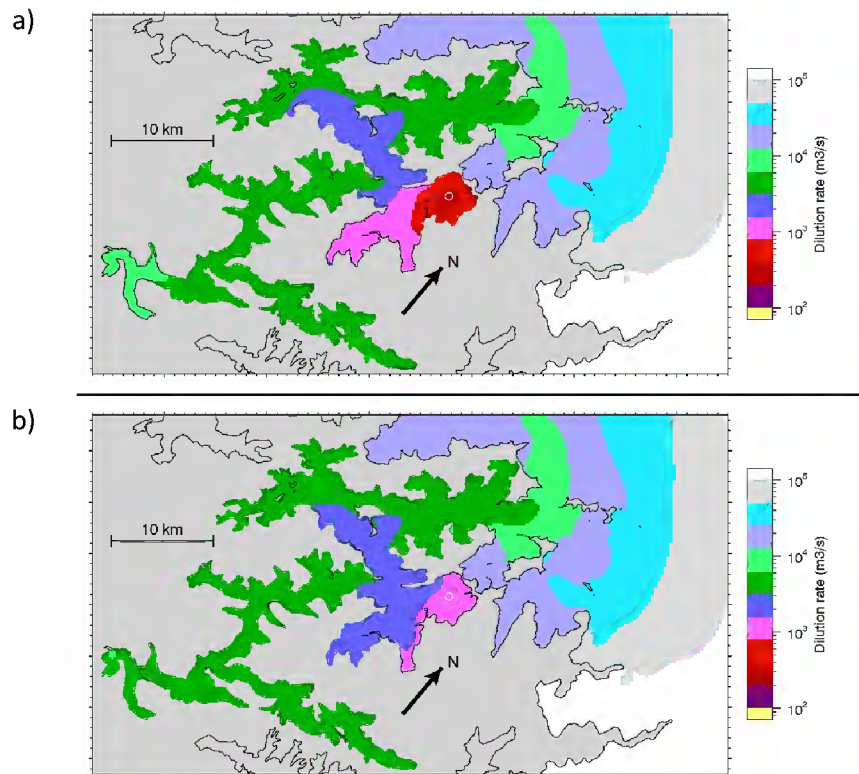


Figure 3-18: Equilibrium concentration for Beatrix Bay tracers. As Figure 3-17 but for tracers released in Beatrix Bay.

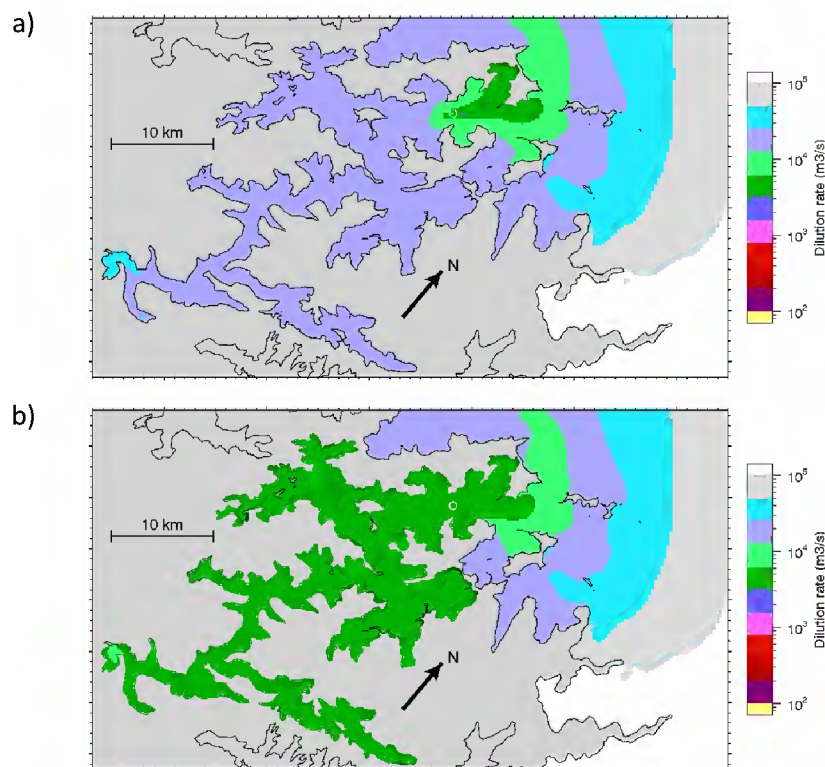


Figure 3-19: Equilibrium concentration for Waitata Reach tracers. As Figure 3-17 but for tracers released in Waitata Reach.

3.7 Hydrodynamic model summary

Grid resolutions from 100 m to 400 m were tested. The 200 m grid reproduces the essential aspects of the hydrodynamics of Pelorus Sound with acceptable accuracy. Comparisons of flushing times derived from the 200 m and 400 m grids show little difference (Section 3.6), which suggests that even the coarser 400 m grid captures the essence of the hydrodynamic behaviour of the Pelorus Sound. The finest grid, 100 m, is too computationally expensive for long simulations (Table 2-1), and in this study is used only for shorter simulations (30 days) for deposition modelling (Section 7).

Comparisons of modelled water levels and currents showed reasonably good agreement with historical field data. For tidal variation in sea level, the model reproduces the observed amplitude for some datasets (the Pelorus Sound 1994–1995 and 1997–1998 measurements) but overestimates it by ~10% for others (the FRIA 2005 ADCP pressure data and the Havelock tide gauge). This indicates that the volume of water moving in and out on each tide is approximately correct, perhaps somewhat overestimated. With regard to tidal currents, the model tends to over predict at some sites, but under predict at others. We note that exact matches between current meter data and model predictions are unlikely due to a number of reasons. In confined waters, small differences in location can result in quite different currents due to the effects of bathymetry (see section 3.3.1). The modelled velocity data are also interpolated from the 200 m model grid onto the location of the current meter, which leads to a degree of smoothing. Comparison of sub-tidal currents (driven by wind and estuarine circulation) from the model and field data show that the model is reproducing the mean currents well, and that the variability in sub-tidal currents is of the correct magnitude, although the timing of fluctuation in the sub-tidal currents do not always agree.

The model's salinity and temperature agree well with observations. The model does tend to under-predict summer temperatures by 1–1.5 °C, however the difference in water temperature between surface and near bed remains about right. This may indicate a deficiency in the model's surface heat flux formulation, which was derived from a global, coarse resolution atmospheric model, or maybe excessive tidal mixing in Cook Strait. However, the model allows stratification to develop to approximately the right extent in Pelorus Sound suggesting that vertical mixing processes are resolved sufficiently well.

The model (and field data) show that the Pelorus Sound is characterised by a strong estuarine circulation with an outward flowing surface layer of brackish water overlying an inward-moving saline water layer. The mean volume flux due to the estuarine circulation (the outward flux in the surface layer) is around $5000 \text{ m}^3 \text{ s}^{-1}$. This compares to peak tidal flows through the Waitata reach of between $20\text{--}30,000 \text{ m}^3 \text{ s}^{-1}$ at neap tide and $50\text{--}60,000 \text{ m}^3 \text{ s}^{-1}$ at spring tide. However the peak tidal flows are sustained for short periods, and reverse regularly, so the estuarine circulation has a very strong effect on the flushing of Pelorus Sound.

Tracer releases to investigate flushing in different parts of Pelorus Sound indicate that the flushing time is of the order of 30–50 days. However there is considerable variability depending on where the tracer is released into the Sound. Substances released near the surface, particularly in the Waitata and Popoure Reaches, are transported outward by the estuarine circulation, leaving the Sound relatively quickly. Substances released nearer the bed are carried inward by the inward moving saline water, before mixing into the surface layer. Thus substances released near the bed will remain in the Sound for longer.

Based on the above information about flows and flushing, we suggest the following idealised picture of transport through Pelorus Sound:

- Transport in Pelorus Sound is driven primarily by estuarine circulation. The dominant supply of freshwater is from the Pelorus River.
- Low river flows lead to a weaker estuarine circulation and therefore longer residence times within the Pelorus Sound.
- Surface salinities decrease (the water becomes fresher) as one moves from outer to inner Pelorus Sound, but the influence of surface freshening events (from increased river flow) occurs through the Sound.
- Stratification in Pelorus Sound is generally driven by salinity. In summer time, when river flows are generally low, warmer surface temperatures can strengthen stratification. In winter, surface salinities can be sufficiently low to allow the surface water to cool to temperature below that of deeper waters.

Possible improvements to the hydrodynamic model could include:

- Generating surface wind fields to drive the model with a higher-resolution atmospheric model.
- Improving the surface-heat flux by using a higher resolution data set.
- Examining and correcting the reasons for suspected excessive tidal mixing in Cook Strait.
- Improved tidal boundary data from a tidal model of Cook Strait rather than the larger-scale NIWA tidal model.

4 Biophysical model: Methods

As described in the introduction, the biophysical model is comprised of several component ‘sub-models’:

- The ROMS hydrodynamic model.
- A so-called nutrient/phytoplankton/zooplankton/detritus (NPZD) model. The particular model that we have adopted includes a simple description of the benthic mineralization of deposited detritus. For that reason, we will refer to it as the *biogeochemical model*.
- A mussel farm model which focuses upon feeding, respiration and excretion.
- A fish farm model which also focuses upon feeding, respiration and excretion.

The hydrodynamic model component has been described in the previous sections. In this section, we describe the biogeochemical, mussel farm and fish farm model components.

4.1 Model description

The ROMS code includes several alternative NPZD sub-models to describe water-column nutrient-plankton dynamics. We elected to base our biological modelling upon the Fennel sub-model (Fennel, Wilkin et al. 2006; Fennel, Wilkin et al. 2008; Fennel, Hetland et al. 2011). We made this choice for the following reasons. Firstly, the Fennel model is one of the simpler biogeochemical models that ships with ROMS. The more complex alternatives will impose an unacceptably high additional computational burden and, in some cases, demand data that are not available for the Pelorus system. Secondly, the available field data would be insufficient to calibrate or validate these more complex models. Thirdly, unlike some of the other sub-models, the Fennel model includes a simple description of benthic mineralization of deposited detritus. Finally, we know that there is a more sophisticated benthic diagenesis (nutrient recycling) sub-model being developed by a group in the USA to accompany the Fennel model. We hope to be able to incorporate that model in the future. Since the Fennel model includes benthic mineralization, we will refer to it as a biogeochemical model.

Regardless of which biogeochemical sub-model is selected, it runs ‘in-line’ with the ROMS hydrodynamic simulation. That is, biogeochemical and hydrodynamic equations are solved simultaneously within the same code-base. The ‘in-line’ approach differs from the ‘off-line’ approach. In the latter, the hydrodynamic model is solved first, and the resulting time-series of water-temperature, salinity, and currents etc., are saved to file with (for example) 15 minute temporal resolution. The ‘in-line’ approach has two great advantages: (a) there is no need to save enormous (100s of GB) files of hydrodynamic results, and (b) the biogeochemical model is able to utilize the fundamental temporal resolution available from the hydrodynamic engine (approximately 12 seconds in our simulations using the 200 m grid).

The Fennel model assumes that nitrogen is the (only) element that might limit biological activity. Field data confirm that nitrogen is the limiting element in the Marlborough Sounds¹⁶. The standard

¹⁶ The term nitrogen limitation implies that concentrations of inorganic nitrogen (primarily NO_3 and NH_4^+) are sufficiently low to constrain realizable individual phytoplankton cellular growth rates more than light intensity (or any other nutrient) does. Theoretically, it is energetically less expensive to synthesize new nitrogenous tissues using ammonium rather than nitrate. Thus, it is common to assume that,

Fennel model that distributes with ROMS has seven obligate state variables (NO_3 , NH_4^+ , small and large (slow- and fast-sinking) detritus, phytoplankton nitrogen, phytoplankton chlorophyll and zooplankton nitrogen) and two optional ones (concentrations of dissolved oxygen and dissolved inorganic carbon). We have added an eighth state-variable (representing very-fast sinking detrital nitrogen stemming from mussel and fish farms – specifically mussel pseudo-faeces, and faeces of mussels and fish)¹⁷. This material is generated only by mussels and fish. In comparison with the other two detrital classes, it sinks very rapidly (5 cm s^{-1} , cf. $0\text{--}3 \text{ m d}^{-1}$ for the other two detrital classes). It mineralizes as readily as the other detrital classes. With the exceptions of the two optional state variables (O_2 , CO_2) and chlorophyll, all the variables are measured in units of nitrogen concentration (mmol N m^{-3}).

The full Fennel model is described in Appendix A. In brief, phytoplankton consume NH_4^+ and/or NO_3 as they grow. Zooplankton consume phytoplankton (and associated chlorophyll). In addition, phytoplankton can die of background processes such as entrapment into small detritus. Large and small organic detritus stems from zooplankton faeces as well as dying phytoplankton and zooplankton. NH_4^+ stems from break-down of the detrital material. In turn, NH_4^+ is oxidized into NO_3 . The chlorophyll to phytoplankton nitrogen ratio evolves in response to the ratio of instantaneous photosynthetic rate relative to the local light-dependent maximum rate. The ratio tends to decline under nutrient-limiting conditions and increase under light-limiting ones. All else being equal, a high chlorophyll content permits greater phytoplankton growth than a low one.

In addition to the explicit coefficients of the Fennel model (Table 10-1), there are some features that are turned on/off by means of pre-processor switches when the model is run. Two of these switches influence the fate of particulate material which settles to the seabed. In our ‘standard’ runs we set them such that 25% of the nitrogenous particulate material which settles on the sea-floor is immediately returned to the water-column as ammonium. The remaining 75% is assumed to be permanently lost through denitrification (Fennel, Wilkin et al. 2006)¹⁸. In our *worst case (no denitrification)* simulations, we set these switches such that all of the sedimenting particulate organic nitrogen would be returned to the bottom-most layer of the water-column as ammonium.

The Fennel model that ships with ROMS does not include mussel farms or fish-farms. NIWA has implemented appropriate mussel farm and fish-farm codes with funding from the Ministry of Business and Innovation and a predecessor body (Foundation for Research in Science and Technology).

The mussel code implements relevant parts of the mussel growth models described in Ren and Ross (2005) and Ren et al. (2010) (with some typographical errors in those papers amended in our code implementation). In particular, the rates of mussel induced particle capture, faecal (and pseudo-faecal) production, NH_4^+ excretion, O_2 uptake and CO_2 production are all incorporated. Mussels are

given the choice, phytoplankton will consume NH_4^+ in preference to NO_3 . When the supply of ammonium is inadequate to meet growth demands, nitrate is used to meet the deficit. Whilst this certainly implies additional energy expenditure there is no reduction of realized phytoplankton growth rates in nitrogen-limited waters. This is because, by definition, the realized phytoplankton growth rate is nitrogen limited – they phytoplankton can accrue more than sufficient (non-nitrogenous) carbohydrates (by photosynthesis) to meet even the elevated energetic demands.

¹⁷ In our earlier (Queen Charlotte Sound and Tory Channel) mode, faeces and pseudo-faeces passed into the already existing ‘large-detritus’ class. Addition of an explicit detrital class for (pseudo-)faecal material was a closing recommendation in Hadfield, Broekhuizen & Plew (2014). The enhancement has been made using NIWA CORE funding from the central Government (project ACEE1502).

¹⁸ The alternative choices were: (a) that the sedimenting material be permanently lost from the system (full denitrification of sedimenting material); or (b) that 100% of the sedimenting particulate nitrogen be instantly returned to the bottom-most layer of the water column as ammonium (no denitrification of sedimenting material).

assumed to have the ability to capture all of the particulate materials in the Fennel model (phytoplankton, zooplankton, small and large detritus). The faeces and pseudo-faeces that they produce pass into a detrital pool that is dedicated to large, very fast-sinking (pseudo-)faecal material¹⁹. The mussel code does not include a dynamic description of mussel growth (biovolume and weight). Instead, the user supplies a time-series of mussel concentration (mussel m^{-3}) for each of several mussel size-classes. A more detailed description of the ingestion/faeces/excretion components of the mussel model is provided in Appendix B whilst section 4.2 describes the manner in which the spatial distribution of the mussel crop was incorporated into the model.

The fish-farm sub-model works in a manner akin to that of the mussel farm. A detailed description of the uptake and release terms stemming from this model is provided in Appendix C. Section 4.3 describes the manner in which the spatial distribution of the fish crops were mapped onto the model grid.

The fish energetics model is based upon that of Stigebrandt (1999). The original Stigebrandt model is designed to conserve energy, carbon, nitrogen and oxygen. It contains descriptions of a maximal size-specific ingestion rate ($\text{J fish}^{-1} \text{d}^{-1}$) from which ingestion (as $\text{g food fish}^{-1} \text{d}^{-1}$) can be calculated using a knowledge of the food composition, faecal production, ammonium production, CO_2 production and O_2 demand. As with the mussel model, we have not implemented the fish-growth component of the model. Instead, the user supplies time-series of fish abundance (fish m^{-3}) for each of several fish size-classes. The user also specifies corresponding time-series of fish feed input rates ($(\text{kg feed/kg fish live weight}) \text{d}^{-1}$) for each fish size-class. If the implied feed input rate ($\text{kg feed m}^{-3} \text{d}^{-1}$) exceeds the implied maximal feed consumption rate, the excess food remains uneaten and its nitrogen content passes into the very fast sinking detritus pool (as do fish faeces).

In the real world, mussels will put on weight over the course of a growth cycle. To achieve that, they must consume more nitrogen than they produce. Thus, in a time-average sense, they are a net sink for environmental nitrogen (though they may be temporary net sources during times when they are receiving insufficient food to offset their respiratory demands). Fish also put on weight over the course of a growth cycle, but they derive their nutrition from an exogenous source (fish feed) rather than from material that is already 'natively' present in the water-column. Any nitrogen that they lose to the environment (faeces and ammonium excretion) augments what is already in the environment. In contrast to mussel farms, fish-farms are a net source for environmental nitrogen.

4.2 Representing the spatial distribution of the mussel crop

Rather than representing each individual mussel line (or mussel farm) as a discrete entity within ROMS, we chose to represent the population of farmed mussels using the grid-structure (spatial resolution) adopted for the ROMS hydrodynamic and water-quality models.

Approximate concentrations of farmed mussels (mussel m^{-3}) within each control-volume of the model domain were derived by adopting several assumptions.

- Since mussel feeding rates etc., are non-linear functions of individual size, we need to prescribe a realistic size-distribution for each population. We know of no data concerning seasonal changes in mussel size structure in the farms within Marlborough Sounds. Thus, we assumed that the size structure remains constant throughout the year, and that all farms share the same size structure. We used four size-classes:

¹⁹ The state-variable is named *XLdetritus* in many of the figures presented later in this report.

32 mm, 47 mm, 72 mm and 100 mm. When required, these lengths were converted to weights using relationships from previous studies (Hickman 1979; Hickman and Illingworth 1980; Orban, Di Lena et al. 2002).

- We assumed that 20% of the length of each dropper was devoid of mussels and that 20% of the length was occupied by each of the four size-classes.
- On the occupied sections of dropper, we assumed that the respective mussel densities for the four size-classes were 170, 150, 130 and 110 mussels per metre length of dropper.
- We assumed that each long line supports 3750 m of dropper per 110 m of backbone (www.NZMFA.co.nz/faq.asp).
- Droppers were assumed to extend from the sea-surface to the lesser of 3 m above the seabed or 15 m below the sea-surface. Time-varying sea-levels imply that the droppers may move into and out of layers of the spatial grid. Almost certainly, the depth to which droppers extend will not coincide with the interface between two model layers. Usually, one intermediate layer (with respect to ordering between sea-surface and sea-floor) will be only partially occupied by the droppers. Thus, the concentration of mussels within each control-volume was recalculated at every time-step of the simulation.
- The mussel farm scenarios were derived from two shape files provided by Marlborough District Council. The first, named "Marine_Farm_Data_13th-Feb2014.shp", contained a series of polygons representing the boundaries of licensed shellfish and fish farms. The second, named "Export_Output_2.shp" contained a series of lines representing mussel farm backbones found in aerial surveys in 2012. All polygons in the first shape file with an approval status of "Active" and a farm type of "Shellfish" were considered to be mussel farms (except that 10 polygons were found to be duplicates and were omitted). We overlaid maps of mussel farms and backbones and assigned those mussel farms that were largely populated by backbones to the *existing farms* category; the remainder were assigned to the *approved farms* category.

Collectively, the above assumptions imply that the size-class specific densities per unit area of mussels within each licenced farming block are 38, 34, 29 and 25 mussel m⁻² for the 32, 47, 72 and 100 mm size classes respectively. Thus the density totalled over all size classes is 126 mussel m⁻².

Figure 4-1 shows the outlines of the *existing farms* and Figure 4-2 the *approved farms*. Black lines in the figures indicate the "Pelorus Sound" region considered for flushing calculations (Section 3.6, Figure 3-12). This region, which includes Forsyth Bay, has an area of 382 km². For the *existing farms* scenario, there are 722 mussel farms within Pelorus Sound, with a total area of 24.44 km². The *existing plus approved farms* scenario adds another 22 farms with an area of 0.69 km², or 2.8% of the *existing farms* area.

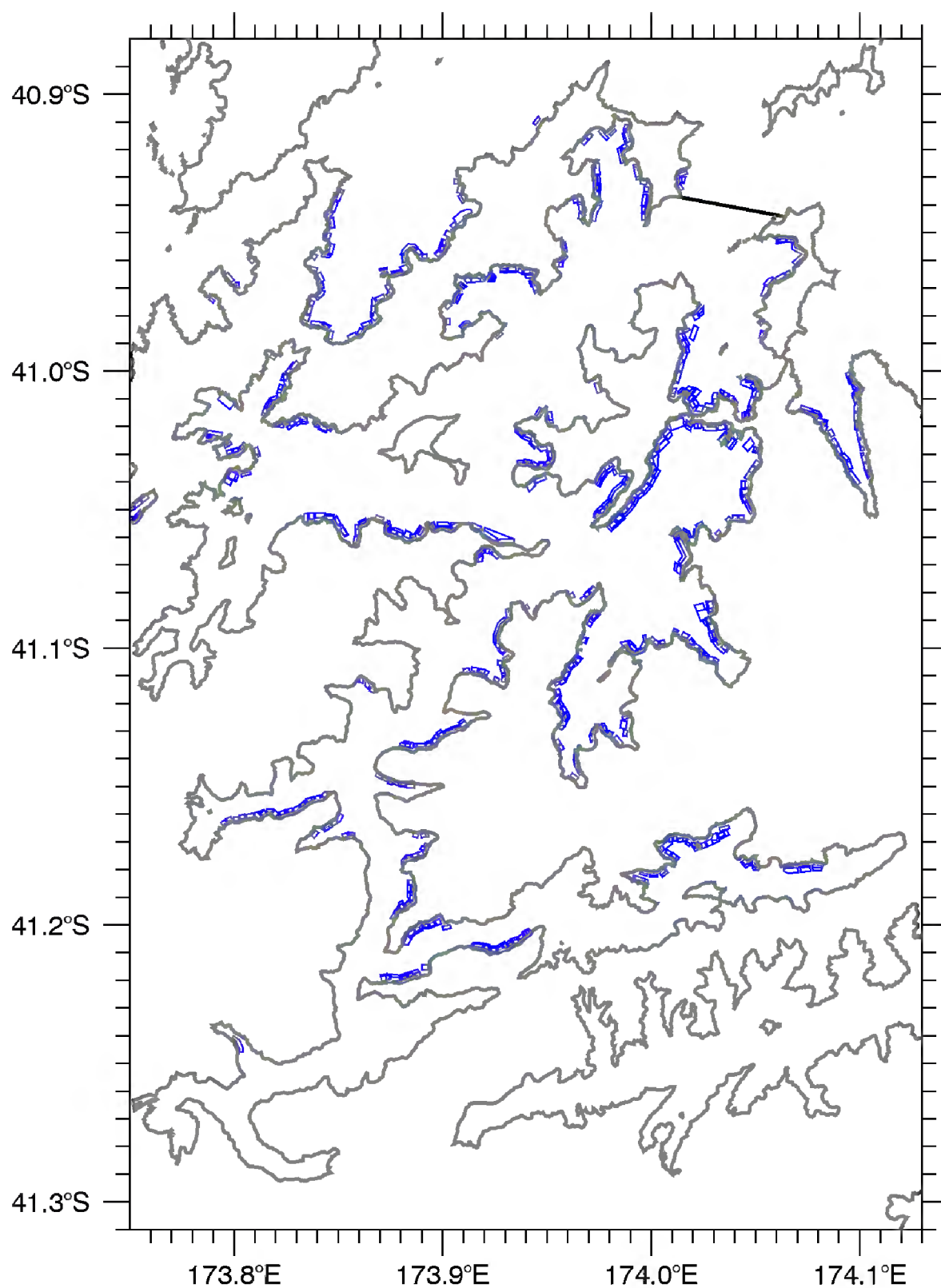


Figure 4-1: Pelorus Sound mussel farm outlines (*existing*). Black lines at the entrance to Pelorus Sound and (barely visible) across Allen Strait indicate the region considered for flushing calculations.

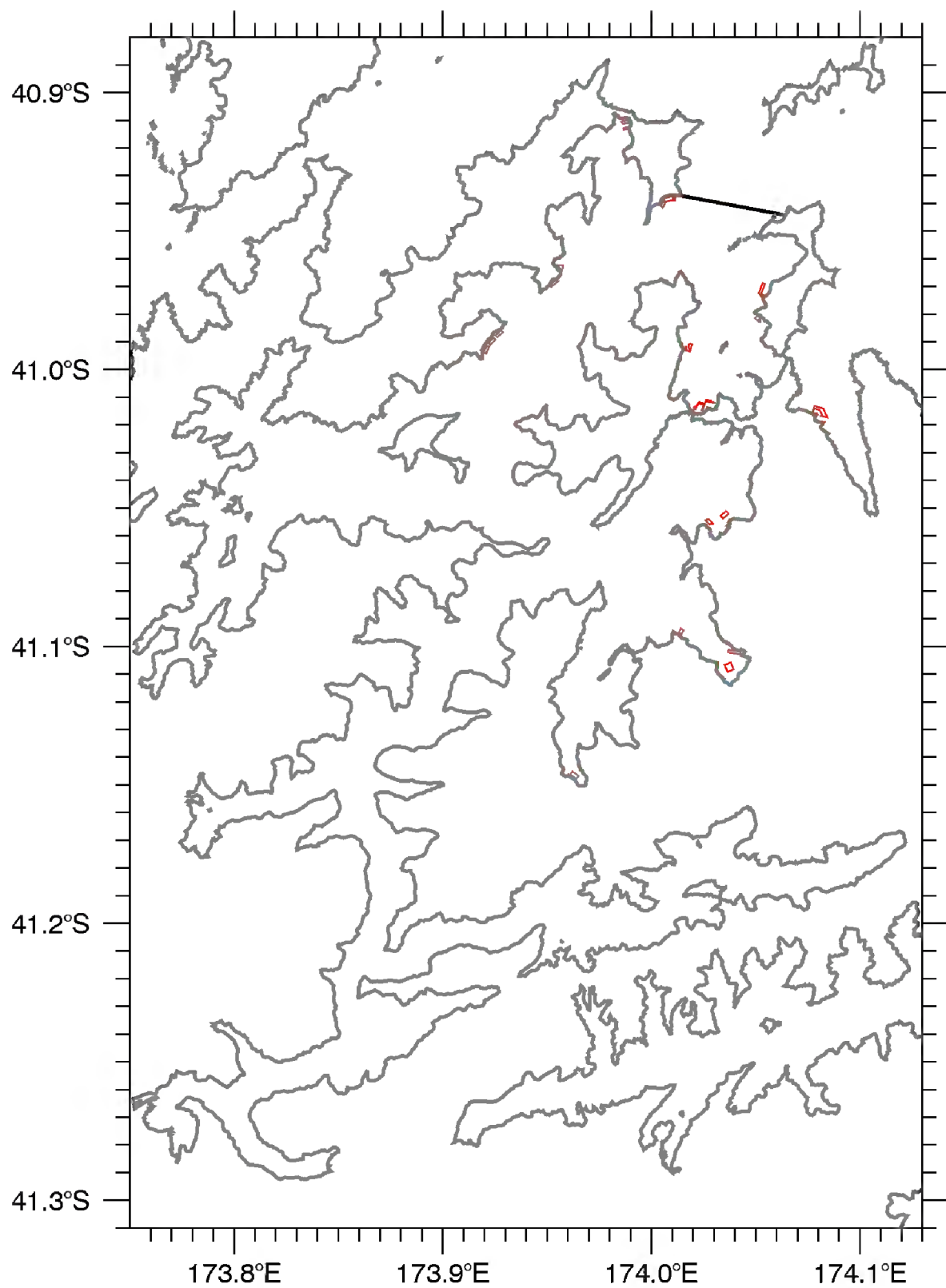


Figure 4-2: Pelorus Sound mussel farm outlines (*approved*).

4.3 Representing the spatial distribution of fish farms

As with the mussel-farms, individual fish-farms were not explicitly represented as discrete entities. Instead, we calculated time-series of farmed-fish concentrations (fish m^{-3}) for each control volume of the model domain²⁰. We did so in a manner similar to that for mussels.

- Marlborough District Council provided us with shape files for each farm. In most cases, these included information on the perimeters of the pens. Where that information was not available, we approached NZKS Ltd. They made their engineering drawings available to us and we digitized the locations of the pen perimeters from these.
- We assumed that cages extend to 20 m below the sea-surface.
- We overlaid the farm perimeters upon the model grid to calculate the fractional area of farm within each water-column of the model grid.
- We assumed that the fish crop associated with each farm was evenly distributed throughout the implied farm-volume (and that feed inputs were evenly distributed across its horizontal surface-area).
- For their existing farms, NZKS also provided us with schedules (time-series) of cohort-and-farm-specific: fish abundance, mean live weight and feed-input rates. We used this information to synthesize farm-specific time-series of: (a) fish abundance (per fish farm) within each of several size-classes, (b) feed input rates (kg feed per kg fish per day by fish size-class). This enabled us to calculate high temporal resolution time-series of population size-structure characteristics and feed input rates that are consistent with the prescribed annual-scale consent conditions and plausible farm management practices.
- For the purposes of this exercise, we partitioned each farm's crop into 14 size classes (individual fish live weight, g): 0–100, 100–200, 200–300, 300–400, 400–500, 500–1000, 1000–1500, 1500–2000, 2000–2500, 2500–3000, 3000–3500, 3500–4000, 4000–4500, 4500–5000.
- We have no detailed information on the proposed stocking- and feeding practices at the new fish farming sites. We calculated hypothetical time-series of feed input, fish density, etc., by: (i) assuming that the farms would operate at their maximum (rather than initial) annual feed input rates; and (ii) rescaling the schedules that NZKS provided for their Te Pangu farm (in Tory Channel) such that the realized annual feed input rates were consistent with the prescribed maximum annual feed input rates²¹.
- For the purposes of modelling, we assumed that each fish farm was entirely enclosed within a single water-column of the model and calculated fish densities accordingly (Figure 4-3). We recalculated the total concentration of fish of each size-class within each control-volume at every time-step (using linear interpolation between the monthly numbers-at-size schedules which we derived from the information provided

²⁰ Given the scarcity of fish-farms within Marlborough Sounds relative to the spatial resolution of the model grid, most control-volumes contain no farmed-fish, and those which do contain farmed fish contain fish from only one farm

²¹ The new farm in Beatrix Bay is not owned by NZKS and will farm Hapuku rather than salmon. Nonetheless, in the absence of any specific information on Hapuku feed-schedules, we applied the rescaled Te Pangu schedule.

to us by NZKS Ltd.). We also calculated control-volume-specific feed input rates at every time-step.

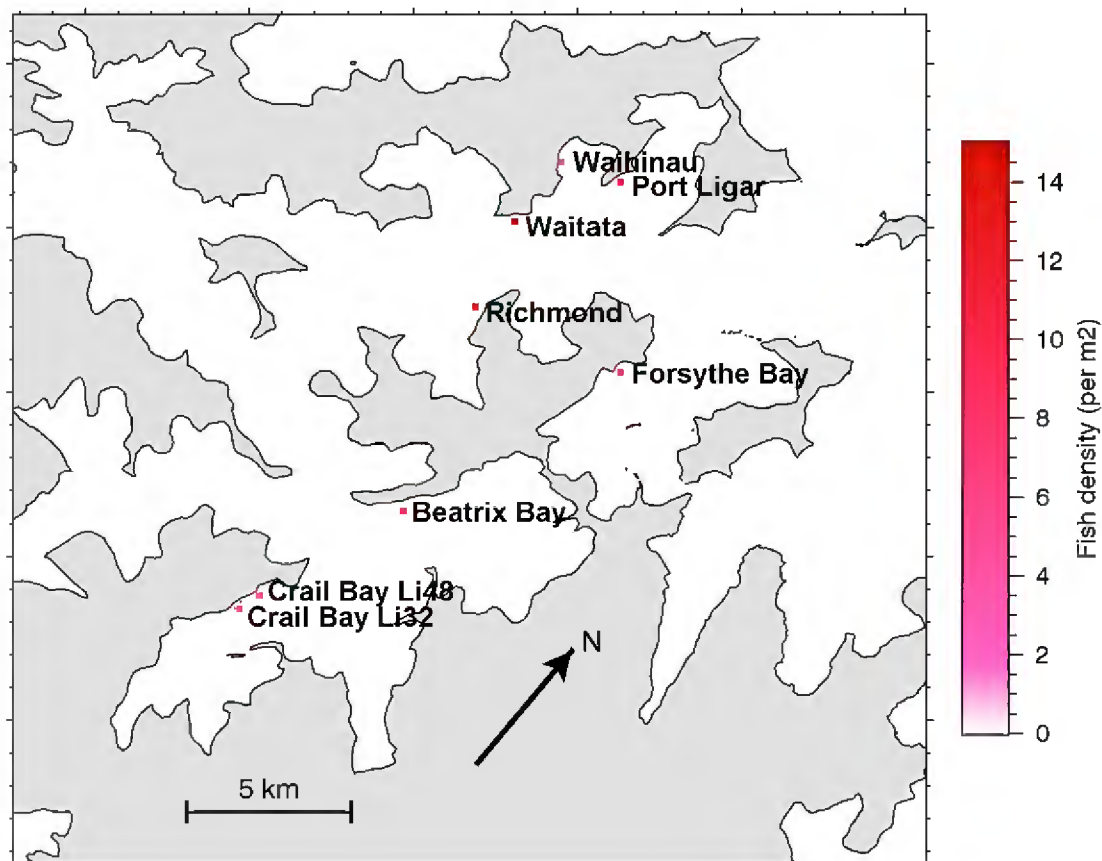


Figure 4-3: Map showing the locations of the eight fish farms. The colour indicates the fish density (fish m^{-2} summed over all of the size classes) averaged over the 200×200 m grid cell around each fish farm during the final 12 months of the simulation. The farms at Crail Bay, Waihināu and Forsythe are the ‘existing farms’. Those at Beatrix Bay, Richmond, Waitata and Port Ligar are the ‘new’ farms. The Beatrix Bay farm is licensed for hapuku rather than salmon but our simulations assume that hapuku feed schedules and physiology will be similar to those of salmon. The farms in Crail Bay are occupied only intermittently, but were assumed to be fully utilized in this modelling.

4.4 Water quality data

Marlborough District Council collect water samples at seven stations (PLS-1 to PLS-7, Figure 4-4). Sampling began in July 2012 and has continued at approximately monthly intervals since then. At each station a near-bed sample is collected from approximately 1 m above the seabed using a bottle sampling device. Up until June 2014 (incl.), the same device was also used to collect a near-surface (approx. 1 m below sea-surface) sample. From July 2014 onward, the near-surface samples have been collected using a hose-sampler that extends from the sea-surface to 12 m below the surface.

Each water-sample was held within an ice-packed chilly-bin and shipped to the NIWA chemistry laboratory in Hamilton within 24 hours of collection. Upon arrival at the laboratory, a small volume of each sample was preserved with Lugols (for subsequent plankton counts). The remainder was frozen until needed for nutrient analysis etc., Table 4-1 provides details of the water-quality variables that

are measured. Quantities measured include: nitrate, ammoniacal nitrogen, dissolved reactive phosphorus, total dissolved nitrogen, total dissolved phosphorus, chlorophyll, suspended solids, volatile suspended solids, particulate carbon, particulate organic nitrogen and counts of phytoplankton and zooplankton individuals by species²². Phytoplankton and zooplankton carbon concentration was derived from the cell counts using measurements of the sizes of individual plankton and published length-weight relationships. In addition, Secchi disk depth, near-surface water temperature and near-surface dissolved oxygen were measured.

Table 4-1: Water-quality variables measured for Marlborough District Council. Phytoplankton and zooplankton counts are made only on the near-surface water samples.

Property	Description	Detection limit	Method or comment
Ammonium Nitrogen	DRP,NH4-N,NO3-N, Simultaneous Auto-analysis	1 mg N m ⁻³	Astoria
Dissolved Reactive Phosphorus	DRP,NH4-N,NO3-N, Simultaneous Auto-analysis	1 mg P m ⁻³	Astoria
Nitrate + Nitrite Nitrogen	DRP,NH4-N,NO3-N, Simultaneous Auto-analysis	1 mg N m ⁻³	Astoria
Volatile Suspended Solids	Filtration, drying at 104 C, followed by furnacing at 400 C	0.5 mg m ⁻³	APHA 2540D
Inorganic Suspended Solids	Filtration, drying at 104 C, followed by furnacing at 400 C	0.5 mg m ⁻³	APHA 2540D
Suspended Solids	Filtration, drying at 104 C, followed by furnacing at 400 C	0.5 mg m ⁻³	APHA 2540D
Turbidity	Turbidimeter rated against Formazin standards	0.1 NTU	APHA 2130B
Chlorophyll a	Acetone pigment extraction, spectrofluorometric measurement.	0.1 mg Chla m ⁻³	A*10200H
Dissolved Reactive Silicon	Molybdosilicate / ascorbic acid reduction.	1 mg Si m ⁻³	APHA4500Si
Salinity	Salinometer, calibrated against seawater standard	0.1 g kg ⁻¹	YSI
Total Dissolved Nitrogen	Persulphate digest, auto cadmium reduction, FIA	10 mg N m ⁻³	Lachat
Total Dissolved Phosphorus	Persulphate digest, molybdenum blue, FIA	1 mg P m ⁻³	Lachat
Particulate Organic Carbon (until June 2014 inclusive)	Catalytic comb @900°C, sep, TCD, Elementar C/N analyser	0.1 mg C m ⁻³	MAM, 01-1090
Particulate Organic Nitrogen (until June 2014 inclusive)	Catalytic comb @900°C, sep, TCD, Elementar C/N analyser	0.1 mg N m ⁻³	MAM, 01-1090
Particulate nitrogen (from July 2014)	Catalytic comb @900°C, sep, TCD, Elementar C/N analyser	0.1 mg C m ⁻³	MAM, 01-1090

²² The counts were made only for the near-surface water-samples. Furthermore, the counts will yield only qualitative abundance information for the larger (scarcer and more mobile) zooplankton.

Property	Description	Detection limit	Method or comment
Particulate carbon (from July 2014)	Catalytic comb @900°C, sep, TCD, Elementar C/N analyser	0.1 mg N m ⁻³	MAM, 01-1090
Phytoplankton abundance	Water samples fixed with Lugols upon arriving at Hamilton labs. Subsequently, cells settled onto graticule slide. Cells within random fields identified (to lowest practical taxonomic resolution), measured and counted under microscope	-	Cell carbon estimated from cell dimensions and taxon-specific conversion factors
Zooplankton abundance	Counted, as for phytoplankton but no size determinations	-	Niskin bottle samples combined with cell counting are not well suited to capturing larger/more mobile zooplankton in sufficient numbers to permit robust abundance estimates. The counts and derived biomass estimates provide only very imprecise estimates of zooplankton abundance.

4.5 Initial conditions

At the start of each simulation, the initial values of all biogeochemical variables were horizontally and vertically uniform at values representative of winter conditions in Pelorus Sound. The flushing time of Pelorus Sound, and the time-scales of other biogeochemical processes operating within the model are such that the model gradually forgets its initial conditions (as it evolves towards a state that is determined by boundary conditions and internal dynamics) within 50-100 days.

4.6 Model coefficients

The coefficients of the Fennel biogeochemical model, the mussel model and the salmon model are listed in Table 10-1, Table 10-2 and Table 10-3. Almost all of the coefficients were left at their default values. Only two were changed from their default values.

The attenuation coefficient for photosynthetically active radiation (PAR) was specifically tuned for Pelorus Sound. We set this coefficient to 0.15 m⁻¹ (based upon measurements of PAR attenuation made during the MDC monthly water quality sampling).

The initial slope of the half-saturation constant for light-limited growth was also changed but we did not tune it to Pelorus specifically. In an earlier exercise (Hadfield, Broekhuizen, Plew 2014b), we had treated this coefficient as a calibration parameter when fitting the model to data from Queen Charlotte Sound. We chose to retain that fitted value for our Pelorus Sound simulations.

The value of one of our two non-standard coefficients is based upon direct measurements of that quantity. The value of the other has not been calibrated to Pelorus Sound data. Thus, we argue that we have made no attempt to calibrate our model to water-quality data from Pelorus Sound. Thus, we argue that the comparisons between simulated- and measured water-quality properties represent validation comparisons rather than calibration attempts. They provide a genuinely independent indication of the model's performance.

4.7 Cook Strait boundary data

There are few historical measurements of water-quality in Cook Strait. Indeed, the only publicly available water quality data that we know of for Cook Strait are those published in Bradford, Lapennas et al. (1986; three summertime surveys during 1980 & 1981). Fortunately, New Zealand King Salmon measured water-quality (nutrients, phytoplankton, chlorophyll, particulate nitrogen, but *not* zooplankton) at five stations around Port Gore monthly from July 2012-April 2013. One of these stations is mid-way across the Port Gore mouth of Cook Strait (Figure 4-4). Earlier numerical modelling (Knight 2012a) suggests that this location will have Cook Strait water-characteristics. Furthermore, the water-quality at this station is markedly different from that of the other four stations (which are well within the bay). It also differs from that of outer Pelorus (PLS7/NZKS7) and outer Queen Charlotte (QCS5) and outer Tory Channel (NZKS22). The nature of the differences are consistent with our belief that the outer Port Gore station is sampling Cook Strait water.

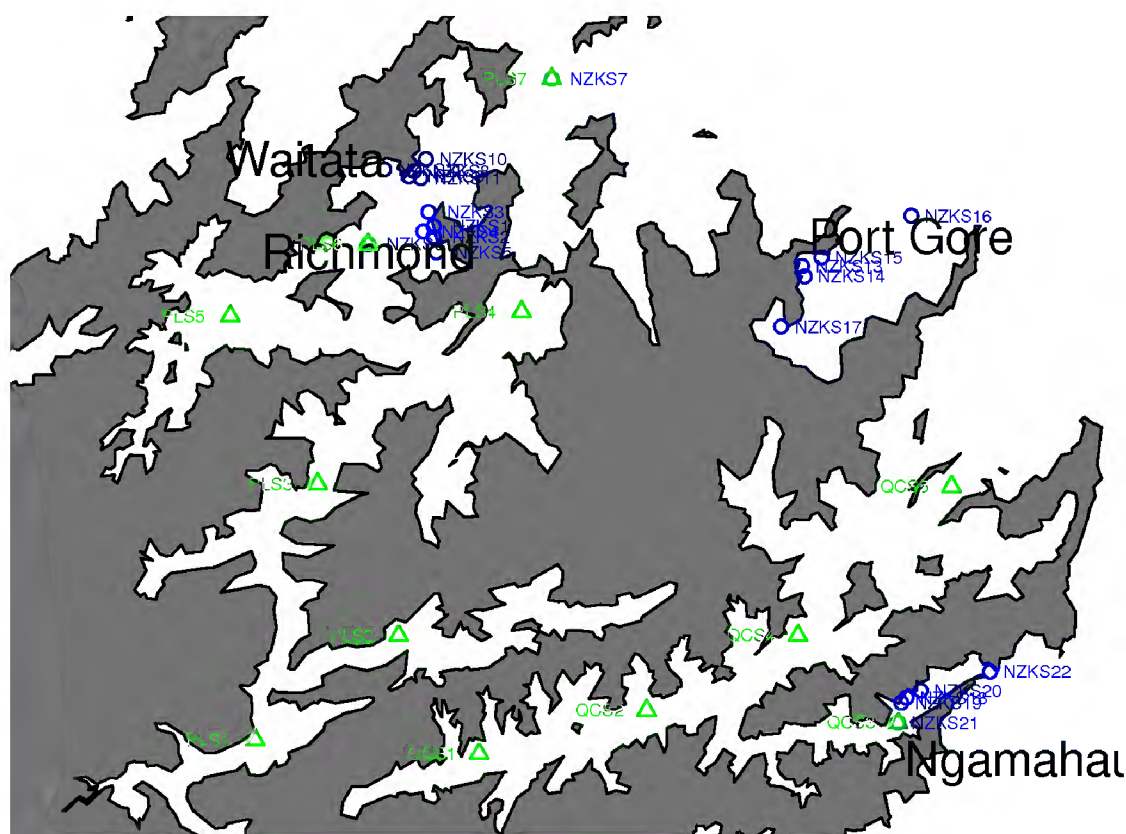


Figure 4-4: Map illustrating the locations of Marlborough District Council (green) and New Zealand King Salmon (blue) water-quality sampling sites. Data from NZKS16 were used to construct the Cook Strait boundary conditions for the NPZD-model.

New Zealand King Salmon Ltd ceased sampling at Port Gore shortly after the Supreme Court upheld the appeal against the Port Gore salmon farm that NZKS had been seeking, but Marlborough District Council continued to sample outer-most Port Gore station (NZKS16) for a further two months. Thus, we have access to one years' worth of monthly data at that station. We generated nominal time-series of sea-surface properties from a 3 month time-centred smoothing curve through the 12 months' worth of near-surface data. We used the corresponding near-bed data to generate a time

series which we assumed to be typical of water at 50 m depth. For the upper 50 m of the water-column, we then used linear interpolation (in the vertical) to derive layer-specific boundary conditions from the smoothed data. Below 50 m, we assumed concentrations were depth invariant (equal to the prescribed values at 50 m).

Zooplankton concentrations have not been measured at the Port Gore station. Thus, boundary conditions for zooplankton were based upon the zooplankton data that Marlborough District Council have gathered at their outer most Queen Charlotte station (station 5). As the zooplankton biomass estimates are imprecise (Table 4-1), we used the time-averaged value from the field data as a temporally invariant boundary condition.

For our modelling, we chose to assume that all dissolved organic nitrogen (DON) is 'old, refractory/inert' material, that is, biologically inactive on the time-scales of interest. Thus, we did not augment the measured NH_4^+ or NO_3^- concentrations so as to generate boundary conditions which implicitly include some reactive dissolved organic nitrogen.

4.8 Catchment boundary conditions

Pelorus Sound has one major river (the Pelorus) flowing into it. Marlborough District Council have collected water-quality samples at a flow recorder station near the Pelorus River mouth at approximately monthly intervals since July 2012 and flow in the river is recorded on a near-continuous basis. They have also gathered similar data near the mouths of three rivers/streams: Kaituna River, Kenepuru Stream, and Cullen Creek. The water-quality monitoring data includes measurements of NH_4^+ , NO_3^- , NO_2^- and total suspended solids.

Concentrations of ammoniacal nitrogen and NO_2^- are negligibly small in comparison with those of NO_3^- . Pelorus river NO_3^- concentrations are not correlated with instantaneous flow, or flow over the preceding 24 hours, but they do show clear annual cycles (being higher in the winter than in the summer).

We adopted a constant boundary condition for ammonium in Pelorus River. For nitrate, we generated a time-varying boundary condition by calculating monthly median values from the Pelorus River monitoring data, and then using linear interpolation to obtain instantaneous concentration values.

Since freshwater phytoplankton and zooplankton will not survive in the salty water of Pelorus Sound, we adopted zero-concentration boundary conditions for chlorophyll and carbon concentrations of phytoplankton and zooplankton. Similarly, we assumed a zero concentration boundary condition for the very large faecal detritus class (mussel and fish faeces).

Concentrations of material that would fall into the model classes 'small detritus' and 'large detritus' have not been measured by MDC. Thus, we must look elsewhere in order to synthesize boundary conditions for these two state-variables. Fortunately, Shearer (1989) reports measured NO_2 , NO_3 , ammoniacal nitrogen (collectively, dissolved inorganic nitrogen, DIN) and total nitrogen (TN) in several streams/rivers that feed into Pelorus Sound. TN-DIN provides a measure of total organic nitrogen (ON). In Shearer's data, the median of the ratio ON/DIN varies between 0.13 (lower Kaituna) and 1.15 (Wakamarina). For the lower Pelorus River, the median ratio is 0.58. We use this last value to derive boundary conditions for in-stream ON concentration (i.e., instantaneous instream $\text{ON} = 0.58 \text{ DIN}$). Finally, we assume that: (i) all of the organic N is bio-available, (ii) that it is all

particulate (such that it can be allocated to either small, or large detritus) and (iii) that it is composed of a 50:50 mix of small and large detritus.

We have chosen to neglect any inputs arising from other (much smaller) point sources such as the Kaituna River, Cullen Creek, Kenepuru Stream and Havelock wastewater plant. We have also neglected any 'diffuse source' inputs that may arise from seeps etc.

4.9 Simulation scenarios

We have made simulations for seven different scenarios:

- No mussel-farms and no fish-farms with benthic denitrification²³ (NM-NF-WD).
- Existing²⁴ mussel-farms, no fish-farms, with benthic denitrification (EM-NF-WD).
- No mussel-farms, existing fish-farms, with benthic denitrification (NM-EF-WD).
- Existing mussel-farms, existing fish-farms, with benthic denitrification (EM-EF-WD).
- Existing+approved²⁵ mussel-farms, existing+approved fish-farms, with benthic denitrification (AM-AF-WD).
- Existing mussel-farms, no fish-farms, without benthic denitrification (EM-NF-ND).
- Existing+approved mussel-farms, existing+approved fish-farms, without benthic denitrification (AM-AF-ND).

We will treat the EM-EF-WD scenario as our 'reference condition' – against which results from alternative scenarios will be compared.

All of the simulations were run on the 200 metre resolution grid. Simulations spanned a 500 day period from 24 May 2012 to 6 October 2013. The EM-EF-WD scenario corresponds to present-day conditions in Pelorus Sound.

4.10 Analysis and presentation of biophysical model simulation results

We made our biophysical simulations on the 200 m resolution horizontal grid. Whilst we have finer resolution grids, the model becomes too computationally expensive to permit annual scale simulations at those finer resolutions (Table 2-1). At 200 m resolution, the detailed structures of individual fish farms and mussel farms are not resolved. However beyond, say, 1 km, natural mixing will have eroded the farm-derived steep gradients to sufficient degree that the grid spacing ceases to be significant. Thus, in the far-field the simulated concentrations will be much less subject to bias. In short, the model has been designed with the intent that it be used to derive an understanding of the regional (and large-bay scale) influences of farming rather than the farm-scale/small bay-scale influences.

²³ such that only 25% of sedimenting N particulate N returns to the water-column as NH_4 , the remainder being lost

²⁴ in this context, 'existing' implies those mussel farms which had lines in the water at the time of a 2012 aerial survey in 2010, and the New Zealand King Salmon fish farms that were operating during the 2012/2013 period.

²⁵ In this context, 'approved' implies those mussel farms which have been approved since the 2012 aerial survey (whether or not they now have lines in the water) and those mussel farms already approved in 2010 which did not have lines in the water at the time of the aerial survey. It also includes the two newly approved NZKS fish farms (Waitata and Richmond) and the small fish farm that has been approved for Beatrix Bay and the Port Ligar fish farm

Simulation results at the locations of each of the seven Marlborough District Council sampling sites within Pelorus Sound were stored at approximately 12 minute resolution. In addition, the 12 hour averaged concentrations for every control-volume were saved once per simulated day.

For the most part, each model state-variable has an unequivocal analogue in the field data, but the situation for model ammonium and model detrital nitrogen is more complex.

Firstly, in reality, non-living organic nitrogen is comprised of both dissolved organic nitrogen (DON) and non-living particulate organic nitrogen (non-living PON). Whilst we have field determinations of DON at the seven MDC sites, the model has no explicit DON pool. Rather a fraction of any newly dead living matter passes into one or other of the two 'standard Fennel' particulate detrital pools (small and large detritus) whilst the remainder passes directly into the so-called ammonium pool. Thus, the question arises: 'how should we apportion real-world DON between modelled ammoniacal nitrogen and the two modelled particulate detrital classes'? Whilst real-world DON concentrations are moderately high (see section 5.2), the majority of marine DON is usually considered to be 'old, refractory' material that is almost inert on the time-scales of interest. We therefore chose to ignore the real world DON when setting our boundary and initial conditions.

Secondly, our direct field determinations of PON measure total (living and non-living) particulate organic nitrogen whereas the model draws distinctions between (living) particulate phytoplankton N, (living) particulate zooplankton N and two classes (small, slow-sinking and large, faster-sinking) of non-living particulate detrital nitrogen.

Plankton nitrogen biomass is known only roughly: from the microscope counts and measurements of individual cells and literature estimates for the volume-specific nitrogen contents of different taxa. Table 4-2 describes the means by which analogues to the model state-variables were derived from the field data.

In short, (i) we assume that field- and modelled ammoniacal nitrogen are direct analogues of one-another, (ii) we derive approximate estimates of living particulate nitrogen from the microscope based counts of phytoplankton and zooplankton and measurements of the dimensions of these plankton, (iii) we use the field determinations of PON as a lower bound for the sum of simulated abundances of large detrital N, small detrital N and living particulate N, (iv) we use the sum of the field determinations of PON and DON as an upper bound for the sum of simulated abundances of large detrital N, small detrital N and living particulate N. Given that we have ignored real-world DON when setting our initial and boundary conditions, we anticipate that the model should yield PON concentrations which are similar to measured PON (rather than similar to the sum of measured PON and DON).

Table 4-2: Means by which the field-data were used to derive analogue values for the model state-values.

Model State-variable	Derivation from field data	Comment
"Nitrate"	$\text{NO}_3^- + \text{NO}_2^-$	
"Ammonium"	$\text{NH}_4^+ + \text{NH}_3$	The model has no explicit DON pool. We choose to lump real-world DON into the model detrital pool (see below)
Chlorophyll-a	Chlorophyll-a	GFC filter (approx. 2 μm pore size)
Phytoplankton carbon concentration	Microscope counts of cells combined with measurements of cell dimensions and literature values for C:volume ratios	The sampling scheme was not designed with zooplankton sampling in mind. The volumes of water that are collected are small. Very motile zooplankton and large jellyfish etc. will be under-sampled. Furthermore derivation of population carbon biomass from cell counts and cell dimensions is error-prone. The zooplankton biomass estimates are certainly very imprecise.
Zooplankton carbon concentration	Microscope counts of cells combined with measurements of cell dimensions and literature values for C:volume ratios	
Total detrital nitrogen (LDetN + SDetN)	(a) PON – phytoplankton N – zooplankton N (b) PON + DON - phytoplankton N – zooplankton N	Given our decision to lump real-world DON into the model detrital pool, (a) & (b) provide lower and upper bounds upon the plausible range of concentrations for the sum of the two model detrital classes. Since we have chosen to ignore real-world DON when setting our initial and boundary conditions, we anticipate that the model should produce PON concentrations that are closer to those of measured PON than those of measured (PON+DON).

For the purposes of illustrating how well the model reproduces the historical field data, we will present time-series plots which show the field data (symbols) and corresponding simulation results (12 minute resolution, from the 'EM-EF-WD' scenario). We will present the results as a series of seven figures. Each figure corresponds to one of the seven Marlborough District Council monitoring stations. Each figure will contain six panels (one each for nitrate, ammonium, chlorophyll, phytoplankton carbon, zooplankton carbon and particulate organic nitrogen). Each panel will show: (a) time-series of field measurements at the near-surface location (red circles), (b) time-series of field measurements at the near-bed location (blue triangles), (c) corresponding simulated time-series at the net-surface (redline) and near-bed (blue line) locations.

We assess the skill (quality with which the model reproduces the field data) by: (a) visual comparison of the results (from the existing-mussels/existing fish-farms/with denitrification simulation) and field data, and (b) calculation of the two skill-measures (bias, B^* , and unbiased root-mean-squared-difference, RMSD'^*) as recommended by Los and Blaas (2010) [who were following Jolliff, Kindle et al. (2008)]:

$$B^* = \frac{1}{N\sigma_D} \sum_{n=1}^N (M_n - D_n)$$

$$RMSD'^* = \frac{sgn(\sigma_M - \sigma_D)}{\sigma_D} \left[\frac{1}{N} \sum_{n=1}^N ((M_n - \bar{M}) - (D_n - \bar{D}))^2 \right]^{0.5}$$

M_n and D_n denote the n^{th} corresponding observations in the model- and field time-series, \bar{M} and \bar{D} denote the means of the two time-series, σ_M and σ_D denote the standard deviations of the time-series.

B^* is equal to the ratio of the difference between the two means relative to the standard deviation of the field data. It is a measure of the degree to which the respective long-term means of the model and field time-series are congruent. Values that are close to zero indicate high congruence. Negative values indicate that the model is, on average, under-predicting relative to the field. Positive values indicate that the model is, on average, over-predicting.

$RMSD'^*$ is a measure of the match between the residuals (after removal of the respective time-series means) in the two time-series. It provides a measure of the degree to which the model reproduces the amplitude and phase of fluctuations in the field data. Like B^* , $RMSD'^*$ is expressed relative to the standard-deviation of the field data. $RMSD'^*$ values which are close to zero indicate that the model is reproducing the amplitude and phase of data-fluctuations well. B^* and $RMSD'^*$ can be plotted against one another in a standard scatter plot. It can be shown (Jolliff, Kindle et al. 2008) that if a point lies within the unit circle centred upon the origin, then the two time-series must be positively correlated. If a point lies outside the unit circle, the two time-series may be either uncorrelated, or weakly correlated; further, the correlation (if it exists) may be either positive or negative. If a point lies outside the unit circle, the simulation time-series explains less of the variance (of the field data) than the simple mean of the field-data does. Thus, points which lie outside the unit circle are indicative of low model skill.

We illustrate the predicted influences which the various alternative scenarios have upon water quality (relative to the EM-EF-WD scenario), in two ways.

Firstly, we will show a series of false-colour figures of time-averaged results. Secondly, we present seven figures akin to the time-series plots described earlier, but in this case, each panel will show five curves (being the simulated near-surface concentrations under five of the simulated scenarios). These figures indicate how instantaneous water-quality at the seven Marlborough District Council stations is predicted to behave under these scenarios. The intent is to demonstrate that the time-averaging employed to generate the false-colour maps is not masking short-lived, but markedly larger, differences between scenarios.

In the false-colour figures, each figure will contain seven rows and each row will contain three panels (maps). Each row of panels corresponds to one of the model state-variables. Each column presents a different *view* (or *property*) of the state-variable. In this context, *view* or *property* is used as a convenient short-hand to refer to: (i, left-hand map of each row) the time-averaged absolute concentration (of the reference scenario) or (ii, central map of each row) relative concentration (alternative scenario relative to reference scenario), or (iii, right-hand map of each row) time-averaged concentration difference (between reference and alternative scenario) for a particular state-variable.

Each panel is a false colour map of the model domain. Pixel colour at any location in the map is indicative of the numerical value of the property in question at the pixel-location (yellow/red being 'high', and blue being 'low'). The following points are worth noting:

- In each individual plot, the colour-scheme has been selected to yield ‘pleasing’ colours that allow the full range of values to be readily distinguished. In many of our images of relative (or absolute) change, even the colours at both ends of the colour-scale correspond to changes that are very small (in relative and/or absolute senses).
- The colours should not be interpreted as indicative of whether or not the magnitude of change might be deemed ‘acceptable’. For example, ‘green’ should not be deemed to imply ‘safe/acceptable’ and ‘red’ should not be interpreted as meaning ‘unsafe/unacceptable’.
- The numerical range spanned by the colour-scale differs for each property that we plot. Thus, when comparing maps of different properties on an individual figure, one must recognise that any specific colour does not necessarily equate to the same numerical value in both maps. Furthermore, even when looking at the same property on different figures, the colour-scales may span differing numerical ranges. A given colour may correspond to a differing numerical value on each of the two panels.

Each row corresponds to a different model state-variable (i.e., ammonium, nitrate, etc.). Within a row, the left-hand most panel will show a time-averaged concentration for the state-variable under a reference scenario (usually, EM-EF-WD). The central panel will illustrate the time-averages of concentration-relative-to-the-reference-scenario (RC_p ; Equation 4 – 1) for an alternative scenario. The right-hand column will show the time-average of concentration difference between the reference scenario and the alternative one. For example, the central column may show results from the EM-EF-ND scenario relative to the EM-EF-WD one and the right-hand panel will illustrate the time-averaged difference between these two scenarios. The time-average of relative concentration is calculated as:

Equation 4-1: Definition of relative concentration

$$RC_p = 1 + \frac{1}{N} \sum_{n=1}^N \frac{P_n^f - P_n^e}{\varepsilon + P_n^e},$$

in which N is the number of time-levels involved in the time-average, while P_n^e and P_n^f represent the simulated 12-hour average concentration P at time-level n in the baseline and alternative scenarios respectively. The scalar ε ($=10^{-100}$) was added to avoid the possibility of a division by zero. RC_p takes the value 1 if the time-average of the differences is zero. If, on time-average, the alternative scenario yields lower concentrations than the baseline scenario, RC_p will take a value less than 1. Conversely, if the alternative scenario tends to yield higher concentrations than the baseline scenario, RC_p will take a value greater than 1. Similarly, a negative time-averaged concentration difference implies that the alternative scenario yields a smaller time-averaged concentration than the reference scenario. A positive time-averaged concentration difference implies that the alternative scenario yields a larger time-averaged concentration than the reference one.

5 Biophysical model: Results

5.1 Existing water quality in Pelorus Sound

In this section, we will introduce some of the field data that Marlborough District Council have collected. Data for some other water-properties will be shown in section 5.2 (in which we compare field data and simulation results).

Figure 5-1 presents the time-series of $\text{NO}_3\text{-N}$ concentrations measured at each of the seven MDC stations. Near surface nitrate concentrations (red symbols) are almost always low at PLS-2 (Kenepuru Sound), PLS-4 (Beatrix Bay) and PLS-5 (Tawhitinui reach). At those sites, they are ‘high’ only for a month or so during mid/late winter. Elsewhere, near-surface nitrate concentrations are moderately high for a larger fraction of the year (but still low during the summer). Maximum annual concentrations tend to be greater in the main-stem of Pelorus (PLS-1, PLS-3, PLS-6) than in the side-arms (PLS-2, PLS-4, PLS-5) or at the Cook Strait mouth (PLS-7). Near-bed nitrate (blue symbols) dynamics are similar to those of the surface layer – albeit that the amplitude of the near-bed annual cycle is smaller than that of the near-surface one at most sites. Whilst near-surface and near-bed concentrations are similar during winter, they tend to diverge through spring and summer. Divergence is greatest in Beatrix Bay (PLS-4), Tawhitinui (PLS-5) and Waitata reach (PLS-6). It is smallest at the two innermost sites Mahau Sound (PLS-1) and Kenepuru Sound (PLS-2).

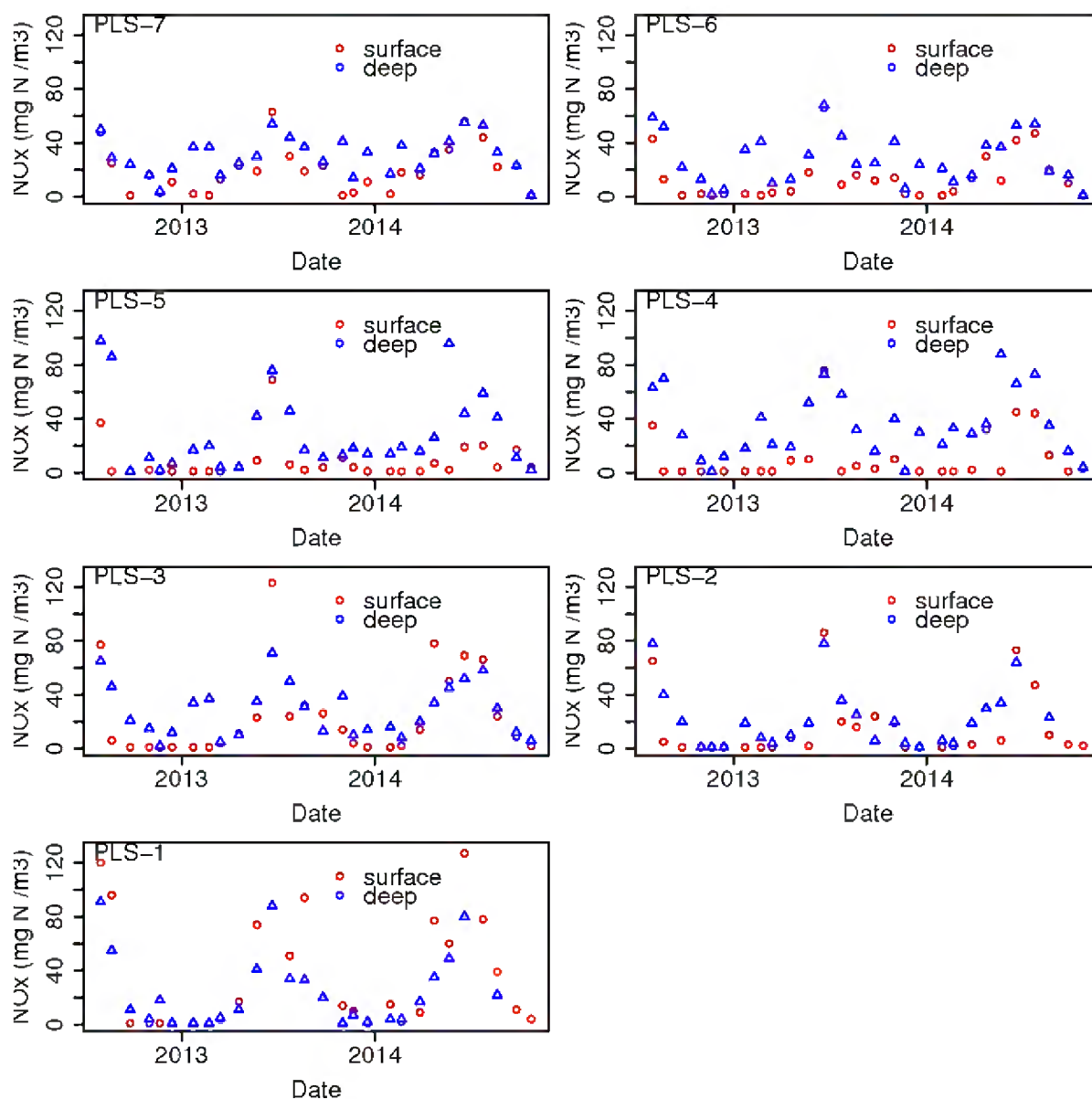


Figure 5-1: Time-series of nitrate concentrations (mg N/m^3) measured at the seven MDC stations in Pelorus Sound.

Figure 5-2 presents the time-series of ammoniacal nitrogen concentrations measured at each of the seven MDC stations. Ammoniacal nitrogen concentrations tend to be highest in mid-late summer. They also tend to be a little higher near-bed than near-surface (Table 5-1). During the summer (December-February, incl.), near-bed and near-surface ammoniacal nitrogen is more abundant than nitrate, but during winter (June – August, incl.) nitrate is more abundant than the ammonium.

Table 5-1: Mean and standard deviation of ammonium and nitrate measured in Pelorus Sound in the MDC sampling. Each observation (value measured at one location in a given month) is treated as an independent record.

Sampling depth	Season	Ammonium concentration mean (standard deviation) [mg N m ⁻³]	Nitrate concentration mean (standard deviation) [mg N m ⁻³]	N
Near surface	Summer	15.3 (8.9)	2.7 (4.9)	434
Near bed	Summer	23.7 (12.2)	17.2 (14.2)	434
Near surface	Winter	12.2 (11.0)	44.1 (37.7)	434
Near bed	Winter	13.0 (9.7)	52.1 (20.1)	434

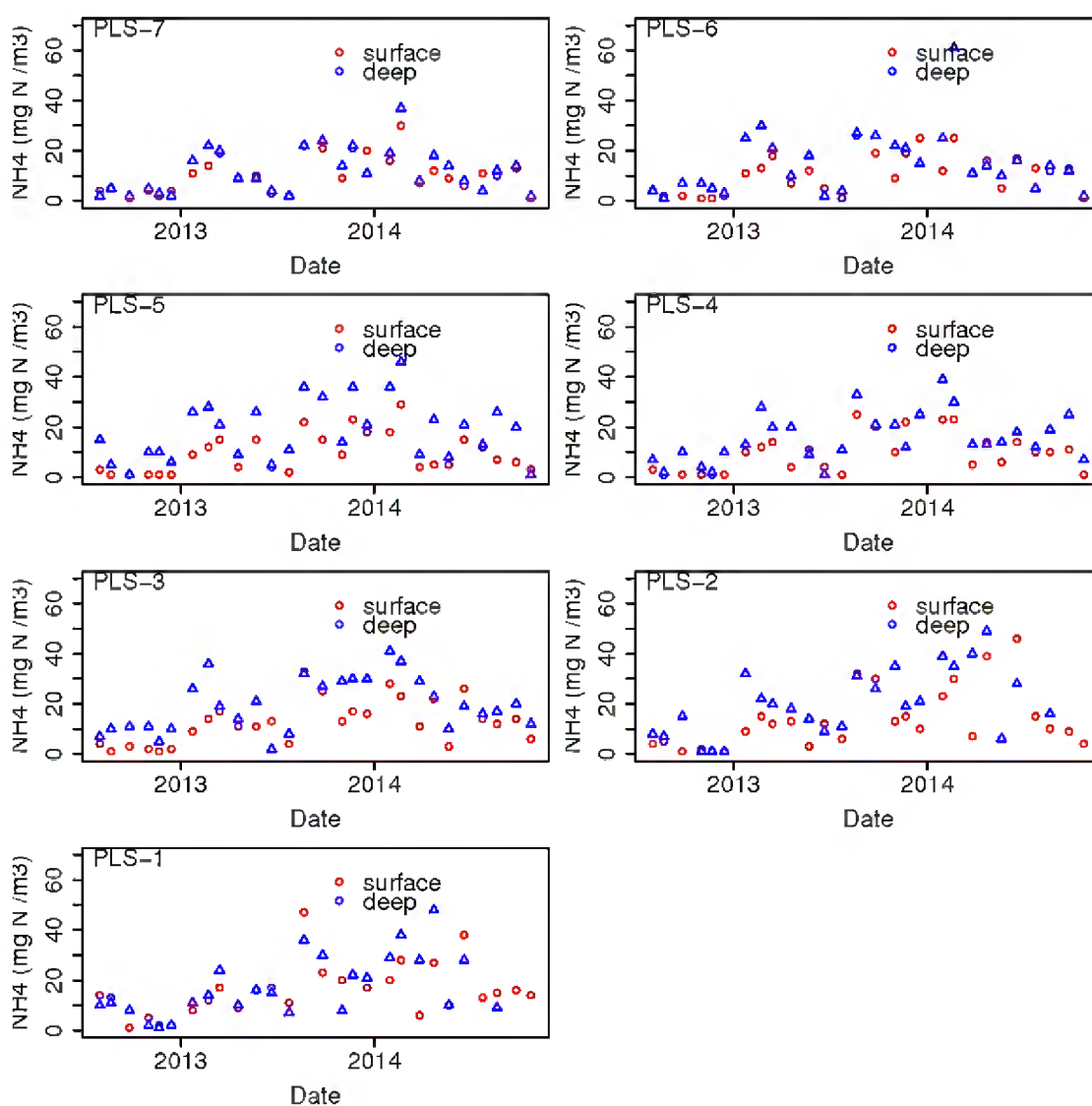


Figure 5-2: Time-series of ammoniacal nitrogen concentrations (mg N/m³) measured at the seven MDC stations in Pelorus Sound.

Figure 5-3 presents the time-series of chlorophyll concentration measured at the seven MDC stations. Chlorophyll concentrations tend to be highest at the two inner-most stations (PLS-1 and PLS-2) and lowest at the two outermost ones (PLS-6 & PLS-7). At most stations, near-surface chlorophyll concentrations usually exceed near-bed ones but the Beatrix Bay (PLS-4) and Tawhitinui (PLS-5) stations often exhibit so-called *deep chlorophyll maxima*: near-bed chlorophyll concentrations were higher than near-surface ones during both summer periods. Chlorophyll concentrations tend to be greatest in late-winter/early spring and late summer/early autumn – however the month-to-month changes in abundance are much less regular than those of nitrate or even ammonium.

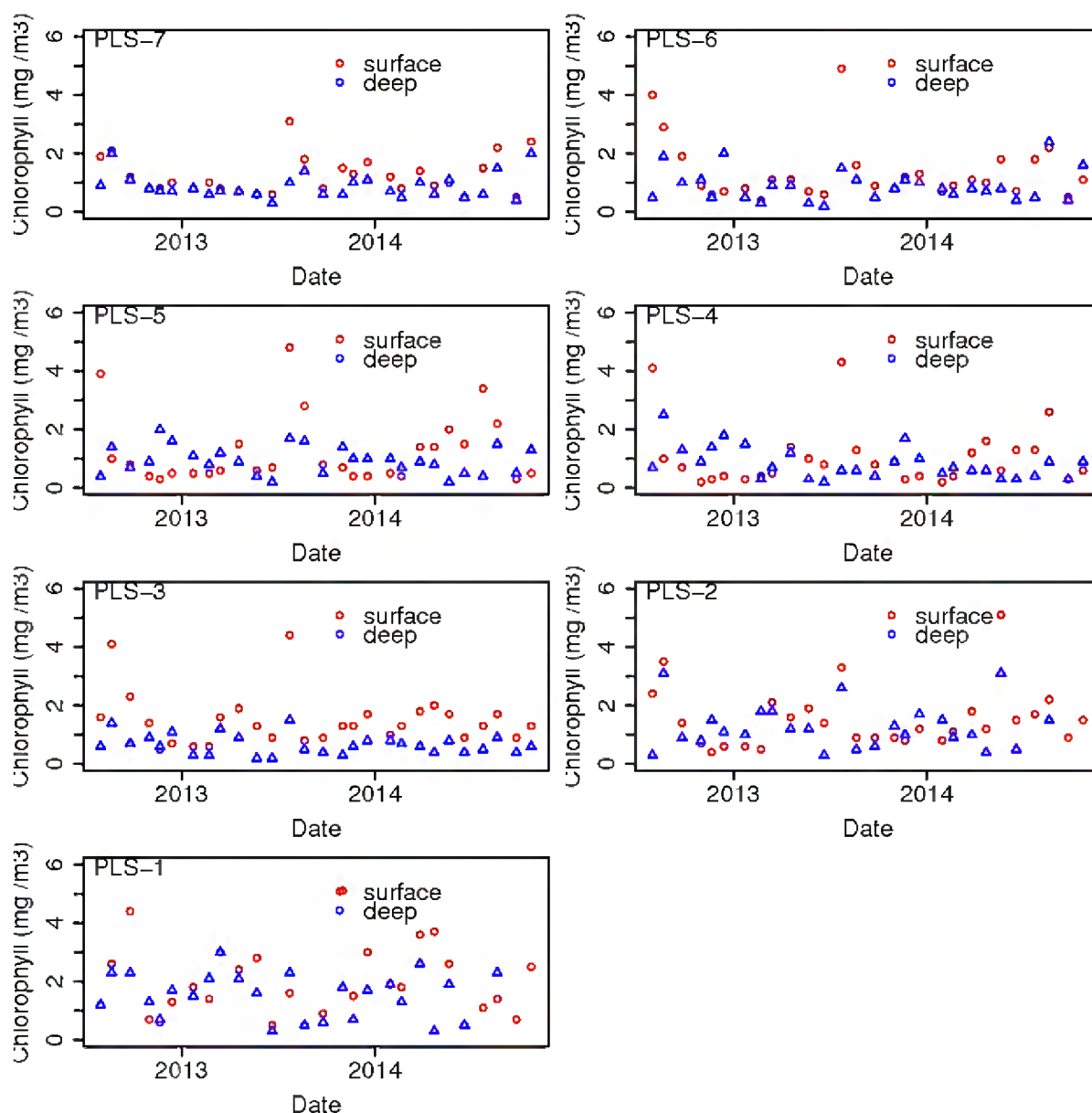


Figure 5-3: Time-series of chlorophyll-a concentrations (mg Chl-a/m³) measured at the seven MDC stations in Pelorus Sound.

5.2 Comparison of simulation results with field data

We have deliberately made no attempt to calibrate the model to any field data from Pelorus Sound²⁶. The coefficients governing the biogeochemical processes are those that we used for the earlier Queen Charlotte modelling (Hadfield, Broekhuizen, Plew 2014b). Since we have not used any Pelorus data to calibrate the model, we can legitimately use the Pelorus data to validate it.

Figure 5-4 - Figure 5-10 present time-series results from the seven Marlborough District Council water quality monitoring stations together with corresponding results from the EM-EF-WD (existing mussel & fish farms, with denitrification) simulation scenario. The congruence between simulation and field measurements tends to be greater at the inner-most (PLS-1 & PLS-2) and outer-most (PLS-7) stations. This probably indicates that: (i) that dynamics at these stations are somewhat constrained/influenced by the boundary conditions which we have applied, and (ii) that the boundary conditions are, indeed 'about right'.

Wintertime maxima- and summertime minima of nitrate concentration are reproduced well at all stations in near-surface waters. Near-bed winter maxima are also replicated well at all stations, but the model under-predicts the extent of the summertime nitrate reduction in the stations that do not lie on the main channel (Tawhitinui & Beatrix Bay). Furthermore, at those stations and also at the two outer stations (PLS-6 & PLS-7), there is a tendency for nitrate concentrations to start rising too early (mid-late summer rather than late-summer-mid autumn). This may be an artefact arising from applying a three-month smoothing window to the Port Gore data from which our boundary conditions were derived.

The field data for ammoniacal nitrogen are less regular than those for nitrate but tend to indicate that ammonium should be more abundant during the summer months and near-bed. The model reproduces those patterns well. The model reproduces the dynamics of particulate organic nitrogen moderately well at all stations – though, perhaps, slightly over-predicting summertime, near-surface PON at stations PLS-3 – PLS-7 and slightly under-predicting the corresponding concentrations in the near-bed water. Jointly, these discrepancies may indicate that our sinking speeds for the small and/or large detrital classes are too low.

Unfortunately, the model appears to over-predict summertime abundances of phytoplankton (as measured by chlorophyll and, more especially, inferred phytoplankton carbon). The over-prediction is worse in the surface layer than the near-bed one, and it is worse in the stations of central Pelorus (PLS-3 – PLS-6) than at the two inner-most stations or the outer-most one. In relative terms, the over-prediction is greater for inferred carbon biomass than for chlorophyll – suggesting that the model is yielding an overly high C:chl ratio. Predicted near-surface PON concentrations are too high during the summer relative to measured PON. Recalling that the model lacks a pool of dissolved organic nitrogen (such that all living nitrogen must pass into PON upon death), it is worth noting that whilst simulated 'PON' exceeds measured PON, it remains well below the sum of measured PON and measured DON. This is consistent with our expectations. Simulated zooplankton concentrations show less variability than is evident in the field data and the simulation does not reproduce the sporadic peaks of zooplankton concentration.

²⁶ Though, as noted earlier, we did adopt a PAR attenuation coefficient that is consistent with values measured within Pelorus Sound.

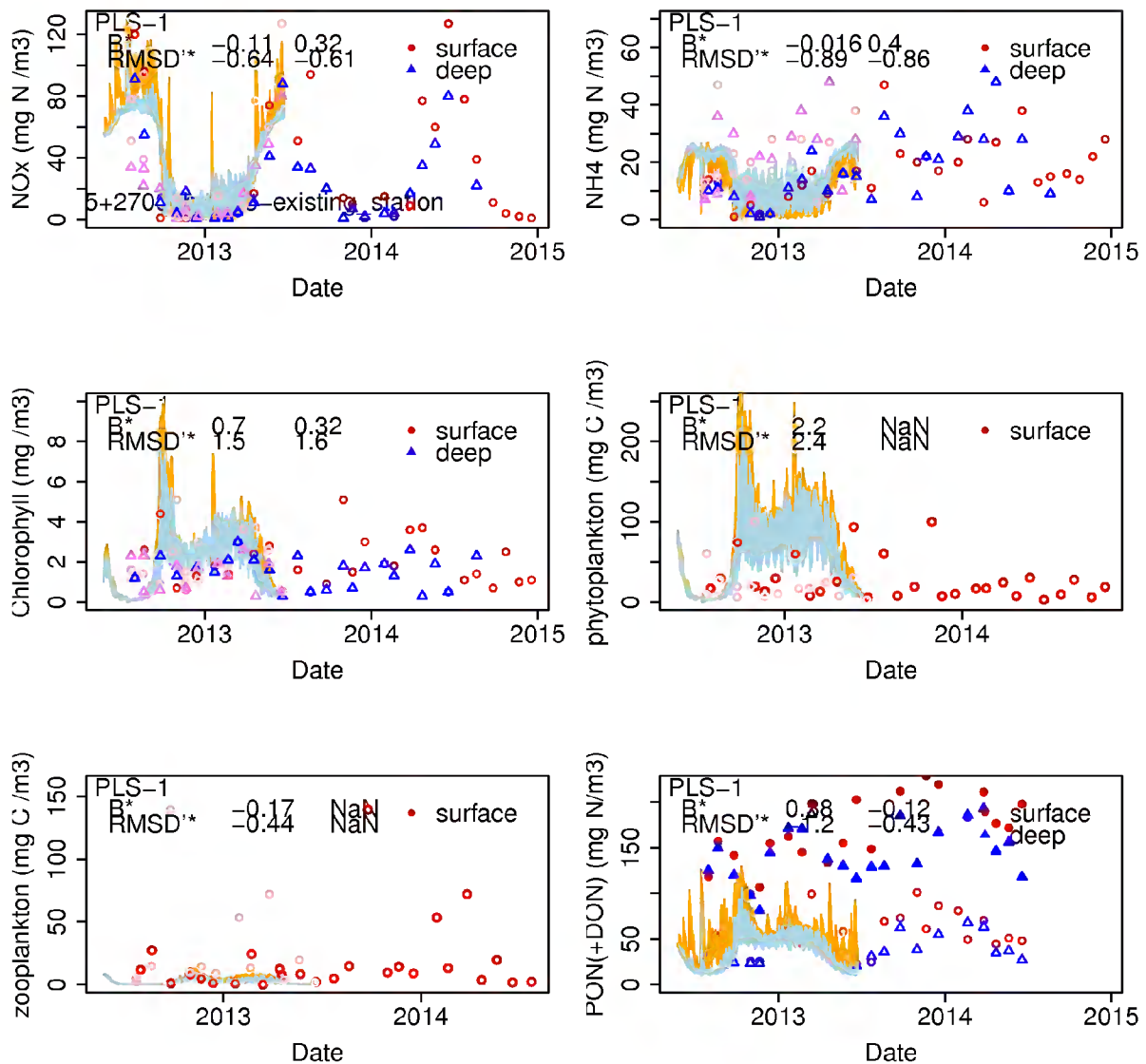


Figure 5-4: Time-series of measured (symbols) and simulated (lines) water-quality characteristics measured at Pelorus station 1. Red symbols are the raw near-surface field-data. Blue symbols are the raw near-bed field data. Pink symbols also represent the near-surface measurements, but in this case measurements made outside of the simulated calendar period have been transposed to a corresponding day-of-year within the simulation period. The violet symbols are the corresponding transposed near-bed field data. The orange and blue lines are the near-surface and near-bed simulation results. Values for B^* and $RMSD'^*$ are also shown. In each case, the left-hand value is for the near-surface simulation/data pair and the right-hand value is for the near-bed simulation/data pair.

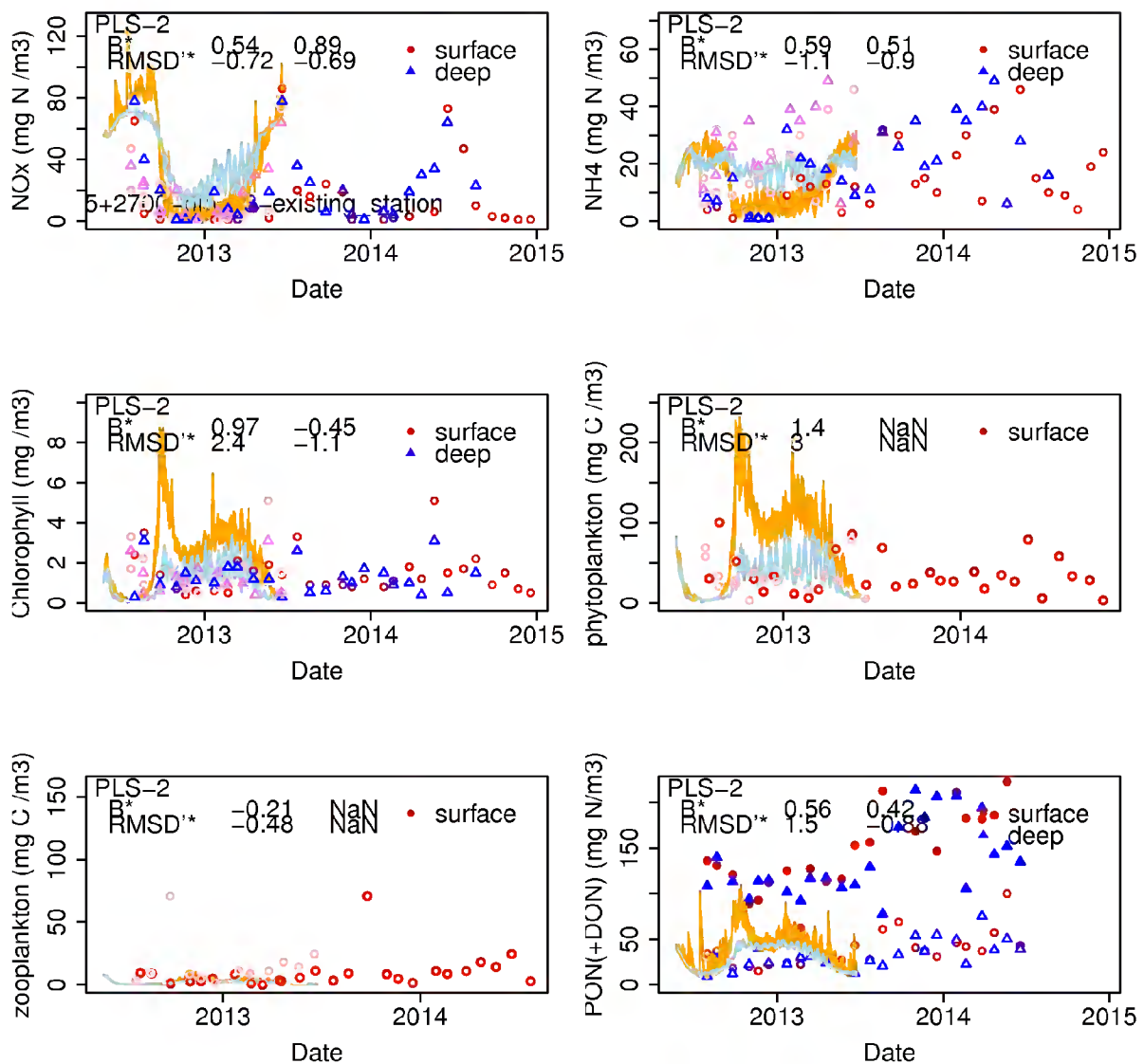


Figure 5-5: Time-series of measured (symbols) and simulated (lines) water-quality characteristics measured at Pelorus station 2. See the caption of Figure 5-4 for further details.

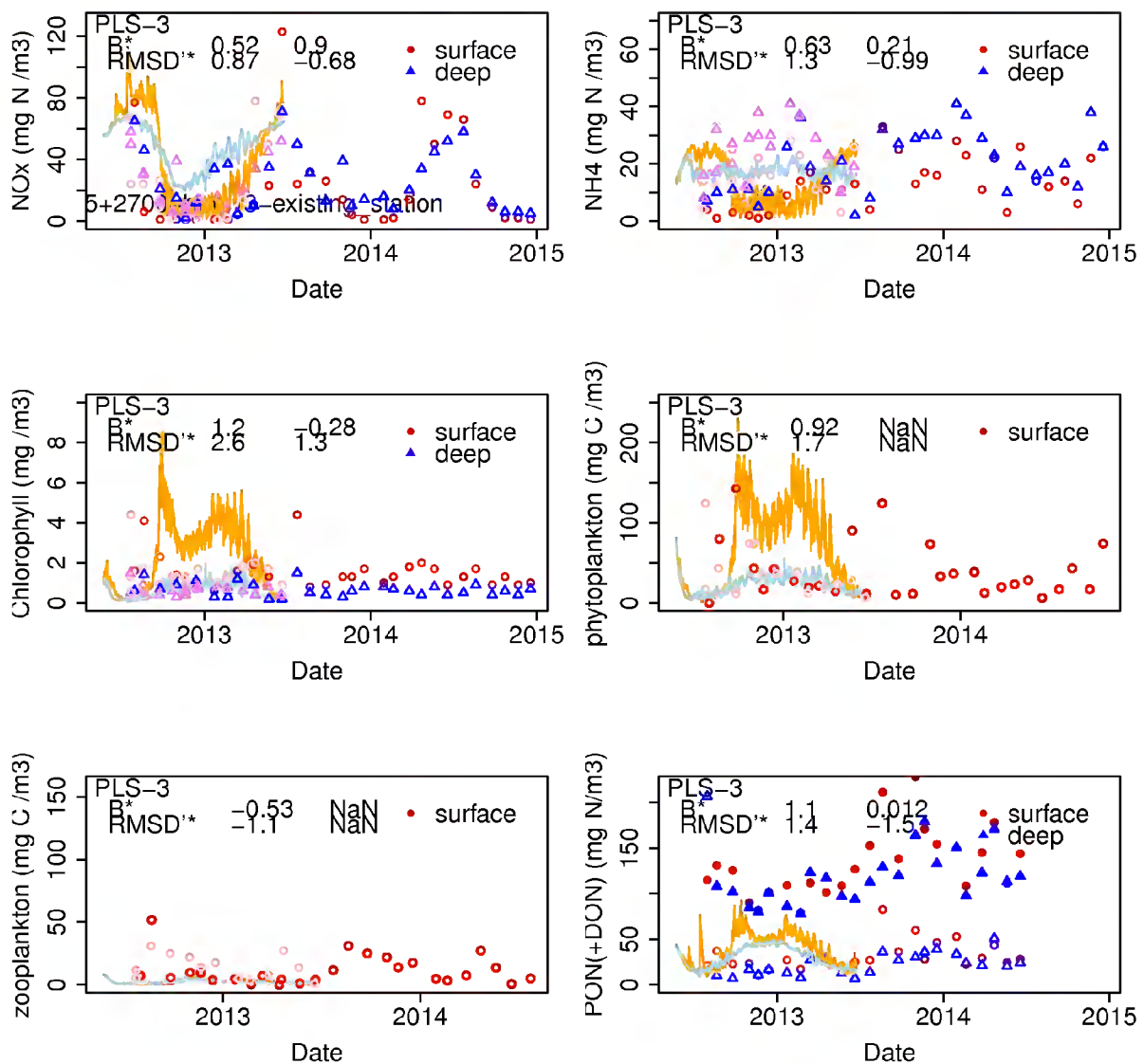


Figure 5-6: Time-series of measured (symbols) and simulated (lines) water-quality characteristics measured at Pelorus station 3. See the caption of Figure 5-4 for further details.

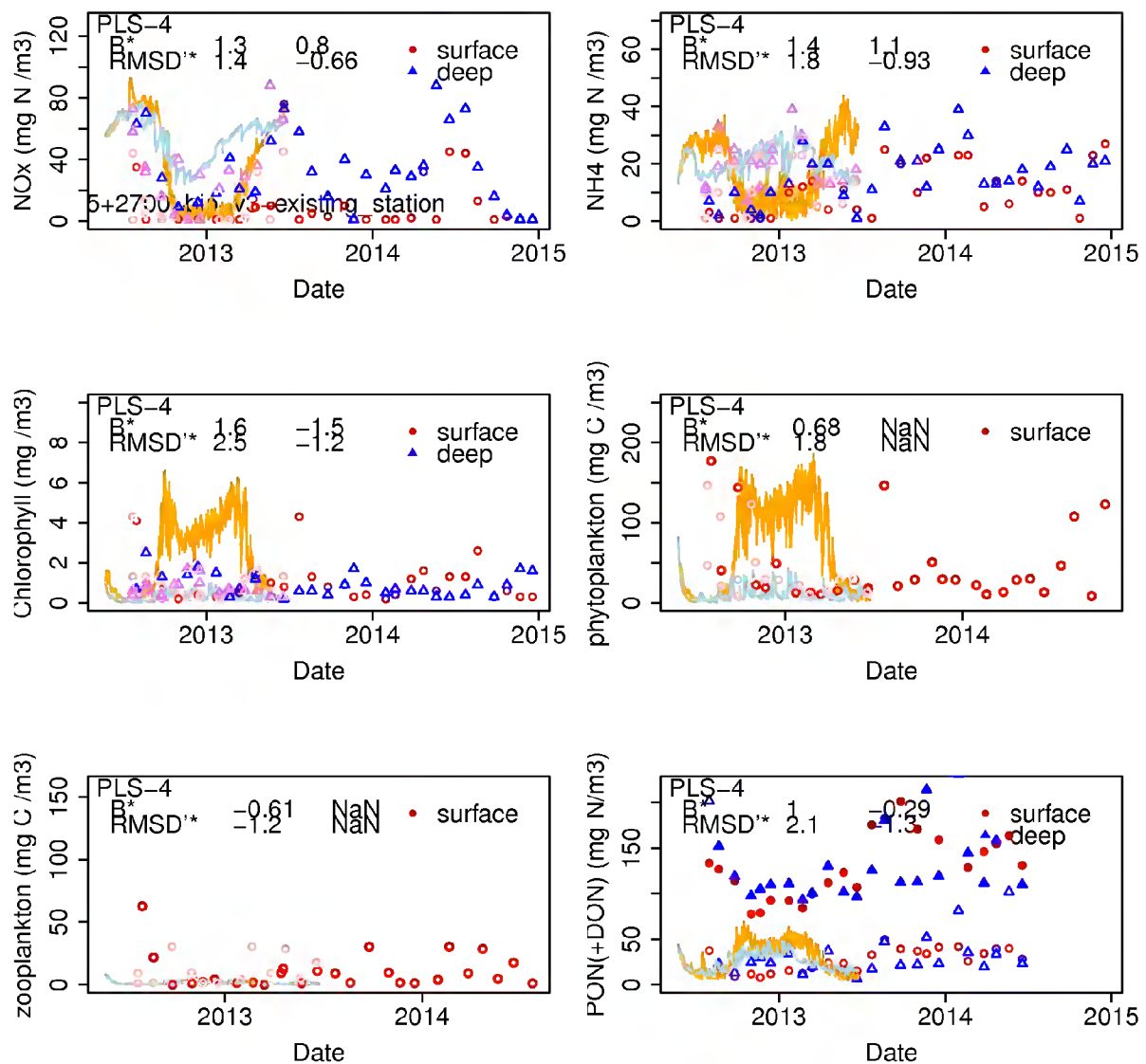


Figure 5-7: Time-series of measured (symbols) and simulated (lines) water-quality characteristics measured at Pelorus station 4. See the caption of Figure 5-4 for further details.

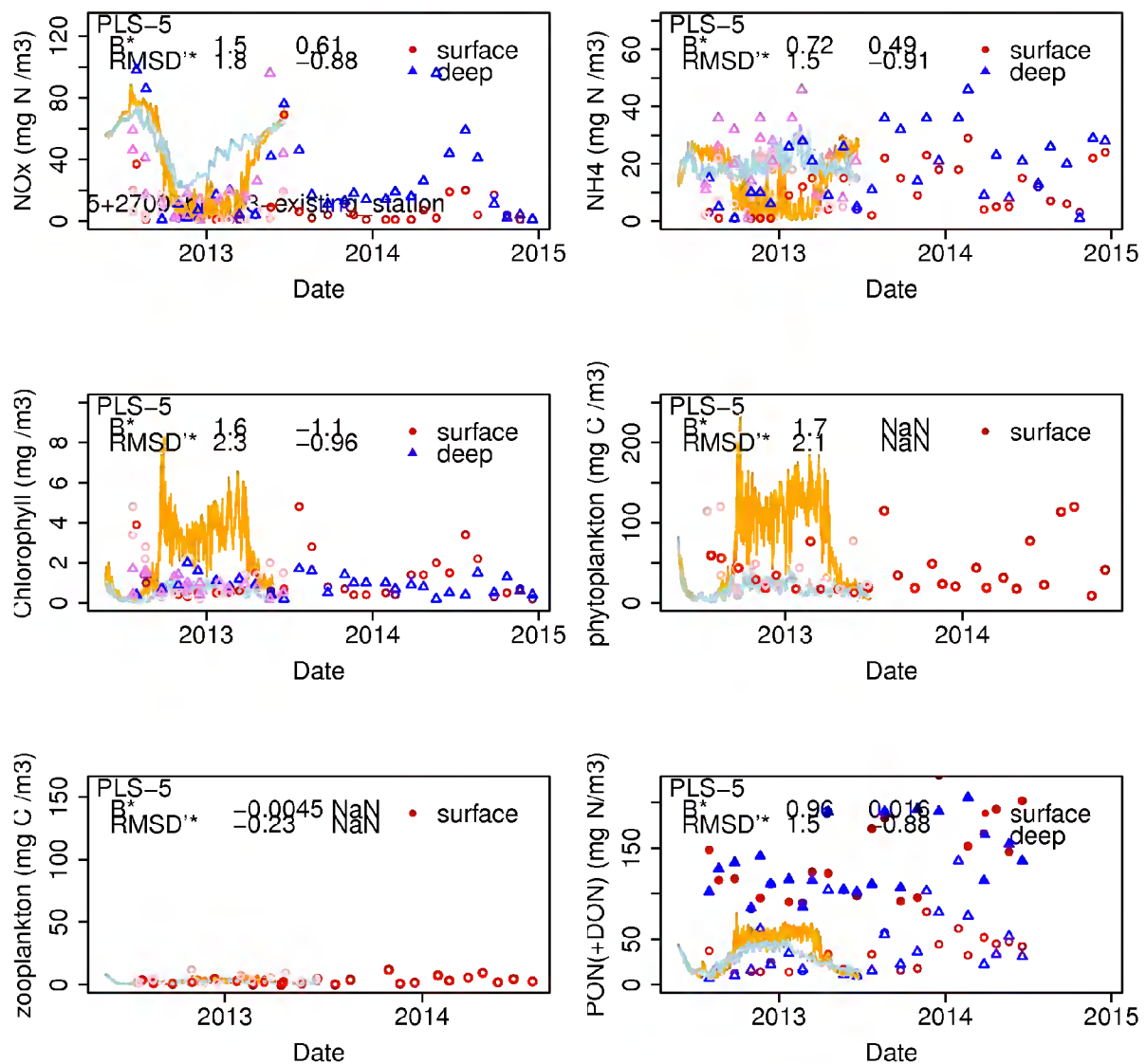


Figure 5-8: Time-series of measured (symbols) and simulated (lines) water-quality characteristics measured at Pelorus station 5. See the caption of Figure 5-4 for further details.

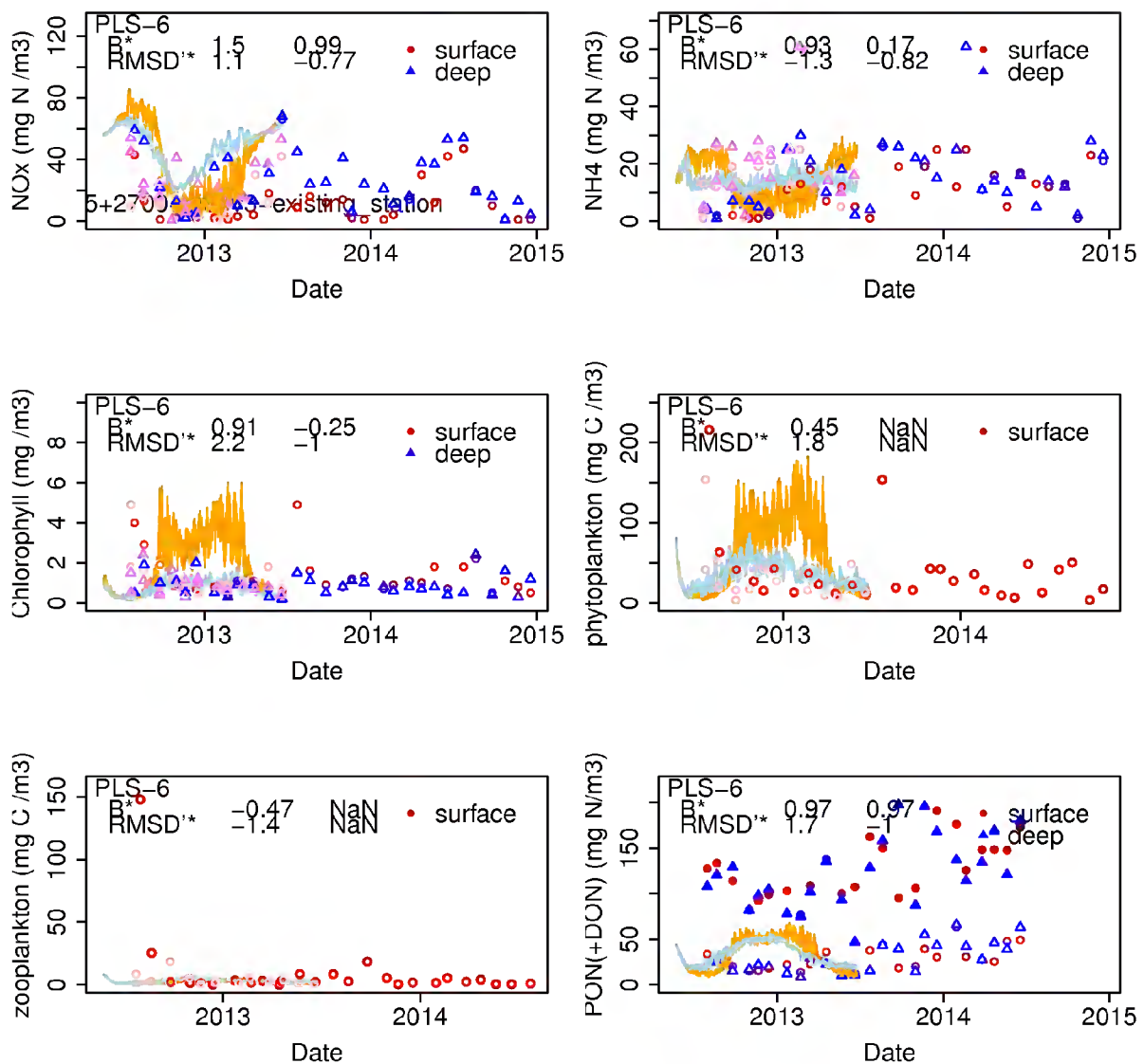


Figure 5-9: Time-series of measured (symbols) and simulated (lines) water-quality characteristics measured at Pelorus station 6. See the caption of Figure 5-4 for further details.

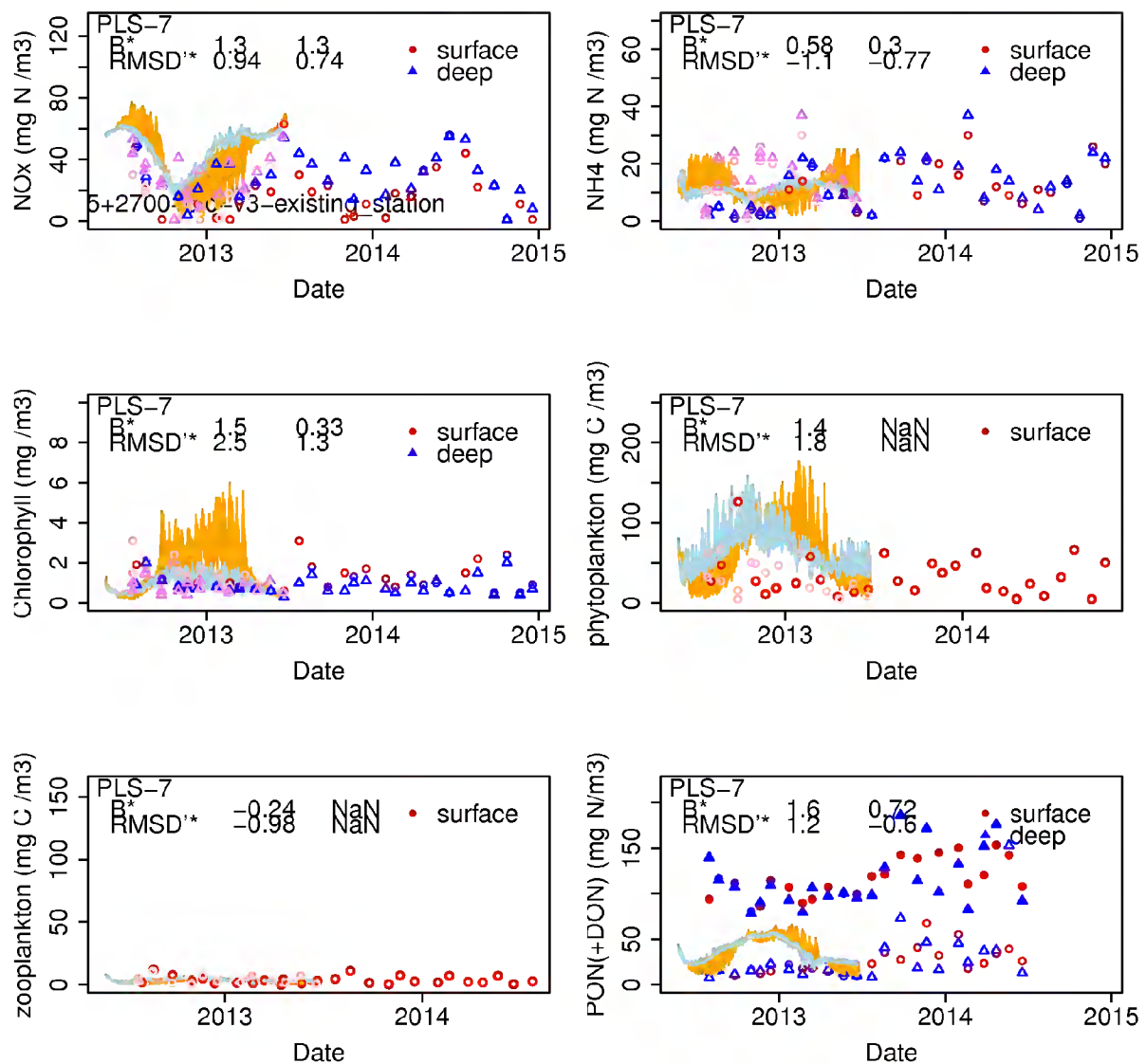


Figure 5-10: Time-series of measured (symbols) and simulated (lines) water-quality characteristics measured at Pelorus station 7. See the caption of Figure 5-4 for further details.

Figure 5-11 presents scatter plots of B^* versus $RMSD^{**}$ for each state-variable. Unfortunately, only a minority of points lie within the unit-circle. Arguably, the model tends to reproduce the near-bed data slightly better than it reproduces the near-surface data. Certainly, it reproduces the apparent phytoplankton dynamics (chlorophyll and phytoplankton carbon) less well than it reproduces the dynamics of other state-variables. We will return to this matter in the discussion (section 6.2).

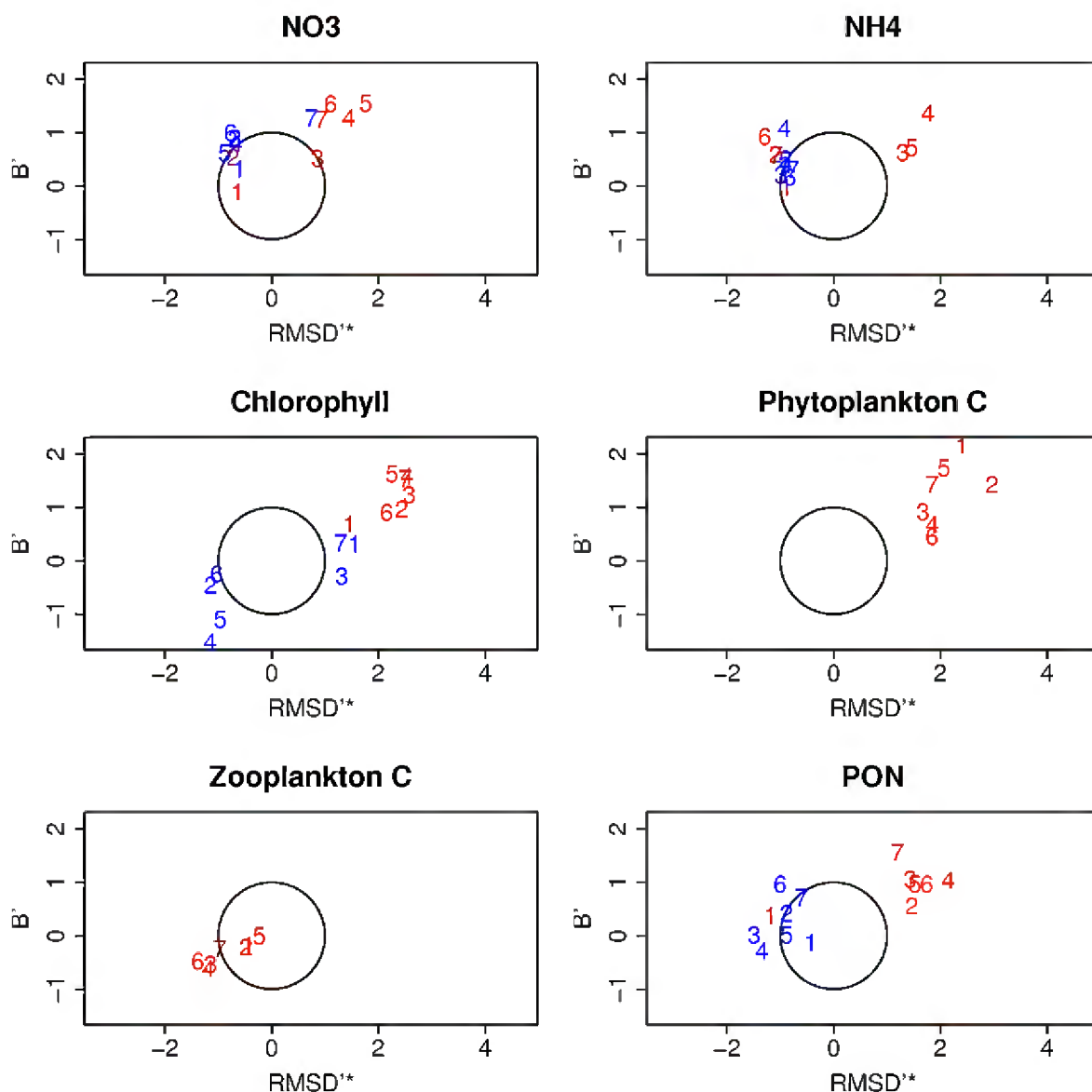


Figure 5-11: Scatter plots illustrating B^* and $RMSD^{}$ for each state-variable for the EM-EF-WD simulation.** Red symbols are for near-surface and blue are for near-bed. The numerals indicate the station number. The unit circle is also shown.

5.3 Denitrification rates

Figure 5-12 illustrates the temporal pattern of denitrification rates under the NM-NF-WD, EM-EF-WD and AM-AF-WD scenarios.

Averaged over the 381 km² of Pelorus Sound, simulated denitrification in the EM-EF-WD scenario rates vary from about 0.7 mmol N m⁻² d⁻¹ in winter up to about 2 mmol N m⁻² d⁻¹ in the summer. These are consistent with measurements made during spring, summer and autumn in Kenepuru Sound (0.7–6.0 mmol N m⁻² d⁻¹ under mussel farms and 0.1–0.9 mmol N m⁻² d⁻¹ at control sites) (Kaspar, Gillespie et al. 1985) and during the summer in Beatrix Bay (0.2 mmol N m⁻² d⁻¹ under mussel lines, rising to 0.4 mmol N m⁻² d⁻¹ at control sites) (Christensen, Glud et al. 2003). Whilst the field data are scarce, and we are comparing Pelorus-wide averages with point values, the comparison suggests that the model is yielding plausible denitrification rates.

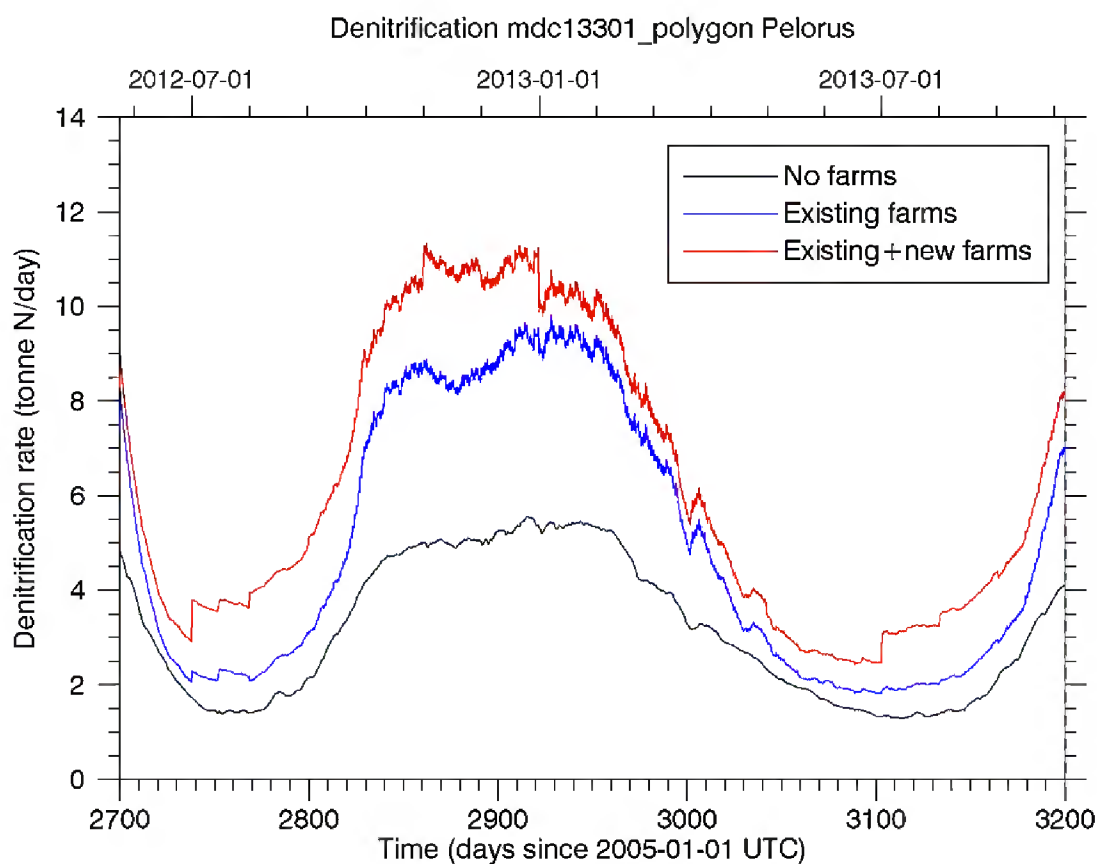


Figure 5-12: Temporal patterns of area-wide benthic denitrification within Pelorus Sound for the NM-NF-WD (black), EM-EF-WD (blue) and AM-AF-WD (red) scenarios.

5.4 Influence of aquaculture and benthic denitrification upon water quality

We start by comparing results from various of the 'existing farm' simulations with the intent of illustrating the relative effects that benthic denitrification, mussel farming and fish farming have upon Pelorus Sound.

The following scenarios were simulated:

- No mussel-farms and no fish-farms with benthic denitrification²⁷ (NM-NF-WD).
- Existing²⁸ mussel-farms, no fish-farms, with benthic denitrification (EM-NF-WD).
- No mussel-farms, existing fish-farms, with benthic denitrification (NM-EF-WD) [at 400 m horizontal resolution].
- Existing mussel-farms, existing fish-farms, with benthic denitrification (EM-EF-WD)
- Existing+approved²⁹ mussel-farms, existing+approved³⁰ fish-farms, with benthic denitrification (AM-AF-WD).
- Existing mussel-farms, no fish-farms, without benthic denitrification (EM-NF-ND).
- Existing+approved mussel-farms, existing+approved fish-farms, without benthic denitrification (AM-AF-ND).

We will begin by comparing the EM-EF-WD simulation results with the NM-EF-WD, EM-NF-WD and NM-NF-WD scenarios. The intent is to determine the sensitivity of the system to the present-day patterns of marine farming. Subsequently, we will compare EM-EF-WD with AM-AF-WD scenario to determine how much the system may change in the future once all existing and approved farms are operating. Finally, we will make two further comparisons. In both, we use a ‘with denitrification’ scenario as a reference, and a ‘without denitrification’ scenario as the alternative. These comparisons are designed to: (i) examine the system’s sensitivity to denitrification, (ii) give an indication of the extent by which the system might change under a ‘worst case’ scenario (in which benthic denitrification ceases) such that the system will become more prone to eutrophication.

We have made comparisons for both the surface-most layer and the bottom-most layer of the discretized domain. Within this section, we focus upon the results from the surface-most layer. Appendix D presents plots that are conceptually equivalent to those presented within this section but which are for the bottom-most (rather than top-most) layer.

5.4.1 “Existing mussel/existing fish/with denitrification” versus “no mussel/existing fish/with denitrification”

Figure 5-13 (winter time-average, surface layer) and Figure 5-14 (summer time-average, surface layer) present comparisons of the EM-EF-WD and NM-EF-WD scenarios. During the winter, the absence of mussels is predicted to induce declines in the concentrations of ammonium (up to approx. 60% decline) and nitrate (up to approx. 30% decline) throughout much of Pelorus. Conversely, concentrations of mussel prey (small- and large-detritus, phytoplankton (as chlorophyll) and zooplankton) tend to increase (up to approx. 2-3 fold increase for detritus and phytoplankton and up to 10-fold increase for zooplankton). Of course, concentrations of XLdetritus (fish faeces and mussel faeces/pseudo-faeces) drop dramatically in the immediate vicinities of each

²⁷ such that only 25% of sedimenting N particulate N returns to the water-column as NH_4 , the remainder being lost

²⁸ in this context, ‘existing’ implies those mussel farms which had lines in the water at the time of a 2012 aerial survey and the New Zealand King Salmon fish farms that were operating during the 2012/2013 period.

²⁹ In this context, ‘approved’ implies those mussel farms which have been approved since the 2012 aerial survey (whether or not they now have lines in the water) and those mussel farms already approved in 2012 which did not have lines in the water at the time of the aerial survey. It also includes the two newly approved NZKS fish farms (Waitata and Richmond) and the small Ngai Tahu fish farm that has been approved for Beatrix Bay.

³⁰ We erroneously included a fish farm at Port Ligar within the ‘approved farms’ scenarios. In reality, the application for a Port Ligar salmon farm has been rejected.

(now absent) mussel farm³¹. The changes (whether decreases or increases) are greatest in the Beatrix/Crail/Clova bays and Kenepuru Sound.

During summer, the qualitative responses of detritus and zooplankton to mussel absence are similar to that of winter: ammonium and nitrate concentrations are predicted to decline whilst detritus and zooplankton concentrations are predicted to rise. The effects tend to be greatest in Beatrix/Crail/Clova and Kenepuru. The summertime response of phytoplankton to mussel removal differs from that seen in the winter. Firstly, the removal of mussels is predicted to have little influence upon concentrations of phytoplankton (chlorophyll) within most of Pelorus (incl. Beatrix/Crail/Clova and Tawhitinui) and to induce a decline in chlorophyll concentration within Kenepuru. Secondly, when mussels are removed, the model shows relative summertime increases of phytoplankton concentration (up to about 30%) at the head of Forsythe Bay, in the western arm of Guards Bay and inner Port Gore that were not evident during the winter.

It is worth noting that, in both winter and summer, simulated concentrations of small detritus (SdetritusN) within Pelorus Sound tend to be higher than those out in Cook Strait, whilst simulated concentrations of large detritus (LdetritusN) tend to be lower than those out in Cook Strait. This is a feature that is repeated in all our scenarios. Our field data measure only total particulate organic nitrogen (PON). We have no way of directly measuring only non-living particulate organic matter³², and no data on the size-composition of this particulate organic matter. When formulating the Cook Strait boundary conditions, we arbitrarily chose to split the PON 50:50 between LdetritusN and SdetritusN. The qualitatively differing spatial gradients of LdetritusN and SdetritusN in our simulations might be evidence that we should have allocated more of the PON into the SdetritusN class.

³¹ This material sinks very rapidly, so even when farms are present, concentrations are negligibly small in those grid-cells which do not contain farms.

³² though we can estimate it by subtracting estimates of phytoplankton and zooplankton biomass from total organic mass

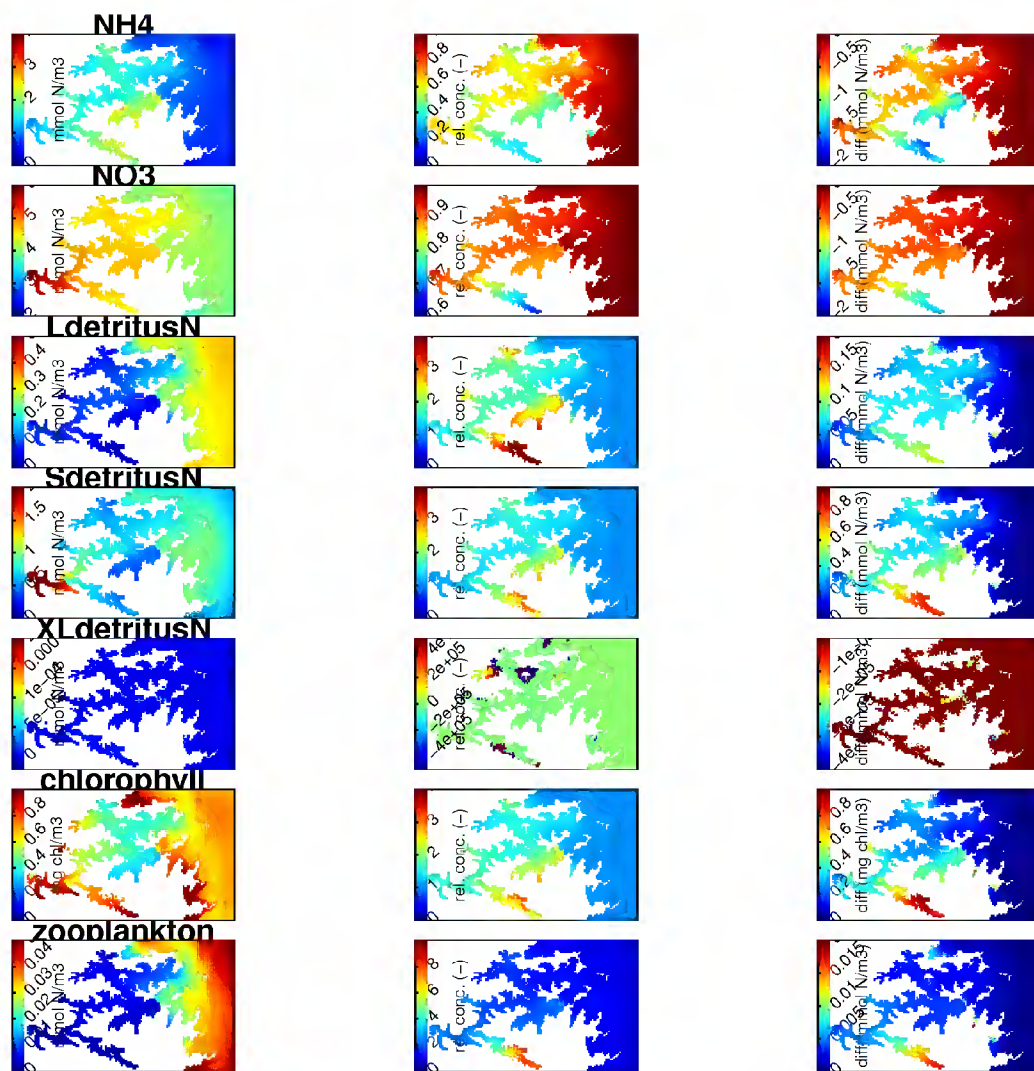


Figure 5-13: Comparison of winter time-averaged surface-layer concentrations in the EM-EF-WD and NM-EF-WD scenarios. The left-hand panel illustrates the time-average in the surface-most layer for the reference scenario (EM-EF-WD). The central panel illustrates the time-averaged relative concentration (alternative scenario relative to reference). The right hand column illustrates the time-averaged concentration difference (alternative scenario - reference scenario). These results are from simulations made with 400 m horizontal resolution.

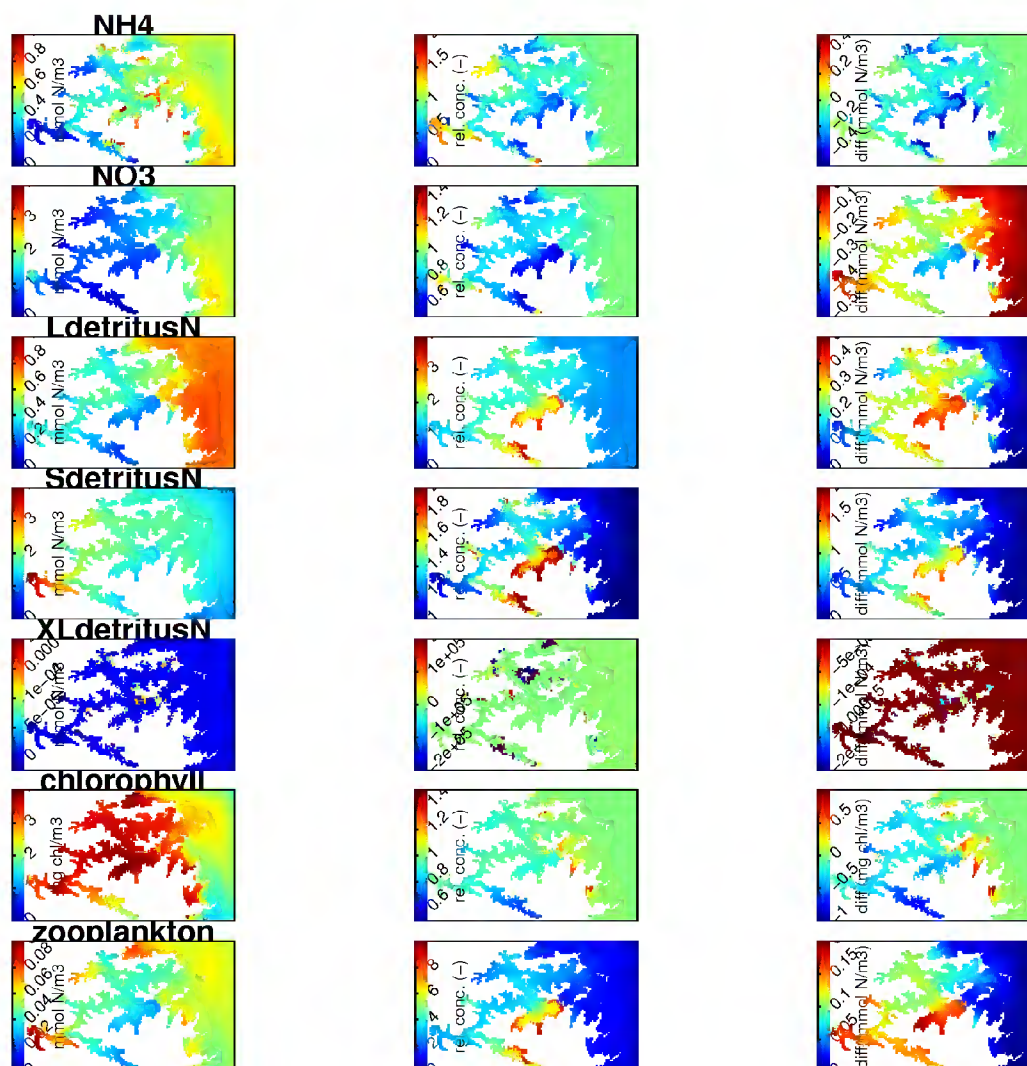


Figure 5-14: Comparison of summer time-averaged surface-layer concentrations in the EM-EF-WD and NM-EF-WD scenarios. Refer to the caption of Figure 5-13 for further explanation. These results are from simulations made with 400 m horizontal resolution.

5.4.2 “Existing mussel/existing fish/with denitrification” versus “existing mussel/no fish/with denitrification”

Figure 5-15 (surface-layer, winter) and Figure 5-16 (surface layer, summer) illustrate the differences between the EM-EF-WD and EM-NF-WD simulations. A casual glance at the colour patterns in the panels might leave the reader with the (false) impression that the existing fish farms are having dramatic effects throughout Pelorus. Inspection of the numerical values on the colour-scales for relative concentration- and for concentration difference will reveal that the magnitudes of change are very small indeed during winter (<2% relative change for all state-variables except ammonium and XLdetritus (mussel and fish faeces and mussel pseudo-faeces³³)). During the summer, the fish

³³ The relative change for the state-variable XLdetritus is very much larger in some places. In those places, the ‘baseline’ concentration of XLdetritus is very small. Despite the ‘large’ (20% or more) relative change, the absolute incremental change is small.

farms do appear to be having bigger effects upon the concentrations of living material and derivative detritus – removal of the fish-farms causes concentrations of SdetritusN, LdetritusN and chlorophyll to drop throughout much of outer Pelorus and Beatrix/Craill/Clova (esp. Craill – which hosts a fish-farm in the EM-EF-WD scenario). The chlorophyll declines are several times greater than those of small and large detritus. The biggest chlorophyll declines are seen in Craill Bay – where they amount to almost 10%. Elsewhere, on Pelorus, they are 5% or less. Zooplankton declines by 5-10% through all of Pelorus and by 10-20% within Craill.

The model suggests that, during winter, mussel farms have a greater impact upon the dynamics of nutrients and seston within Pelorus Sound than fish farms do. During the summer the effects of fish farms and mussel farms are of more similar absolute magnitude (but, at some places and/or times, of differing signs to those associated with mussel farming).

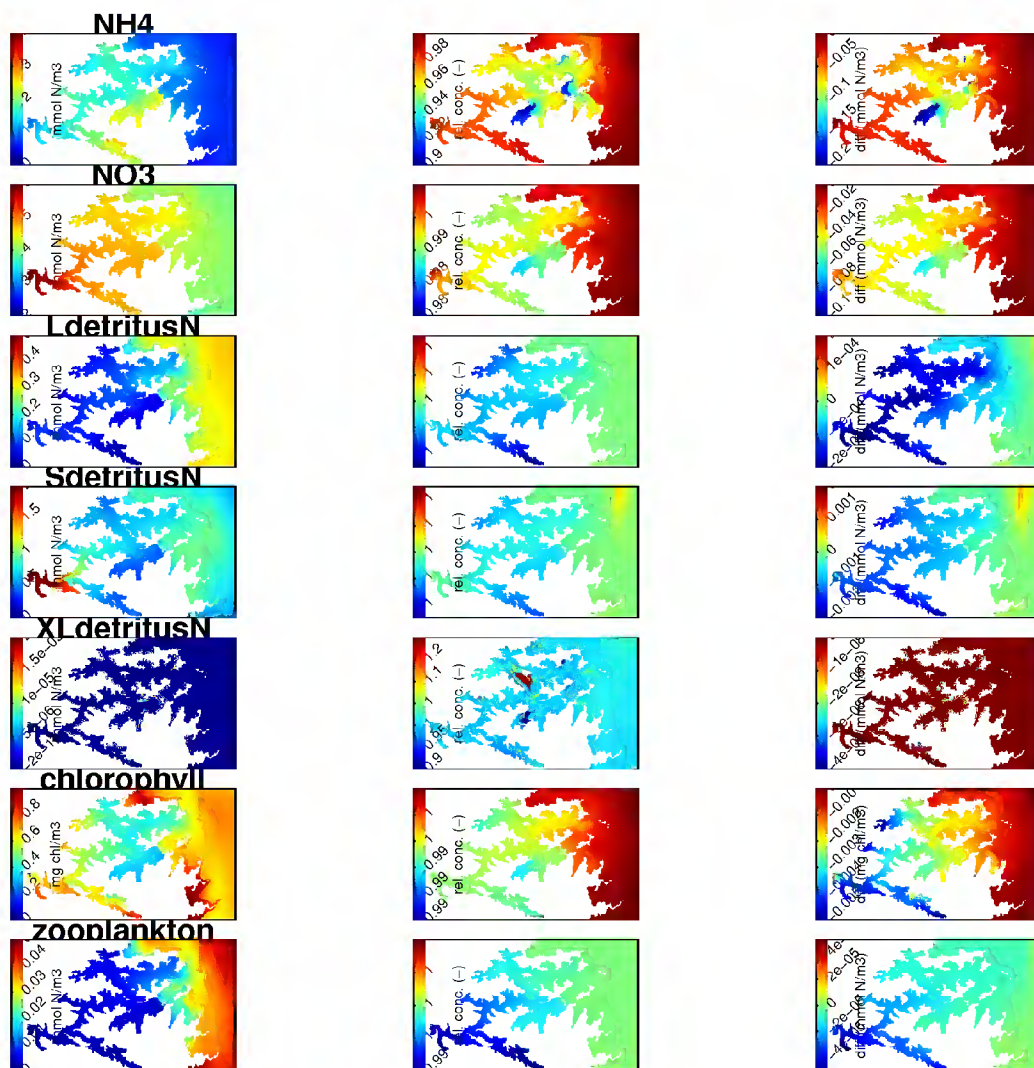


Figure 5-15: Comparison of winter time-averaged surface-layer concentrations in the EM-EF-WD and EM-NF-WD scenarios. Refer to the caption of Figure 5-13 for further explanation. These results are from simulations made with 200 m horizontal resolution.

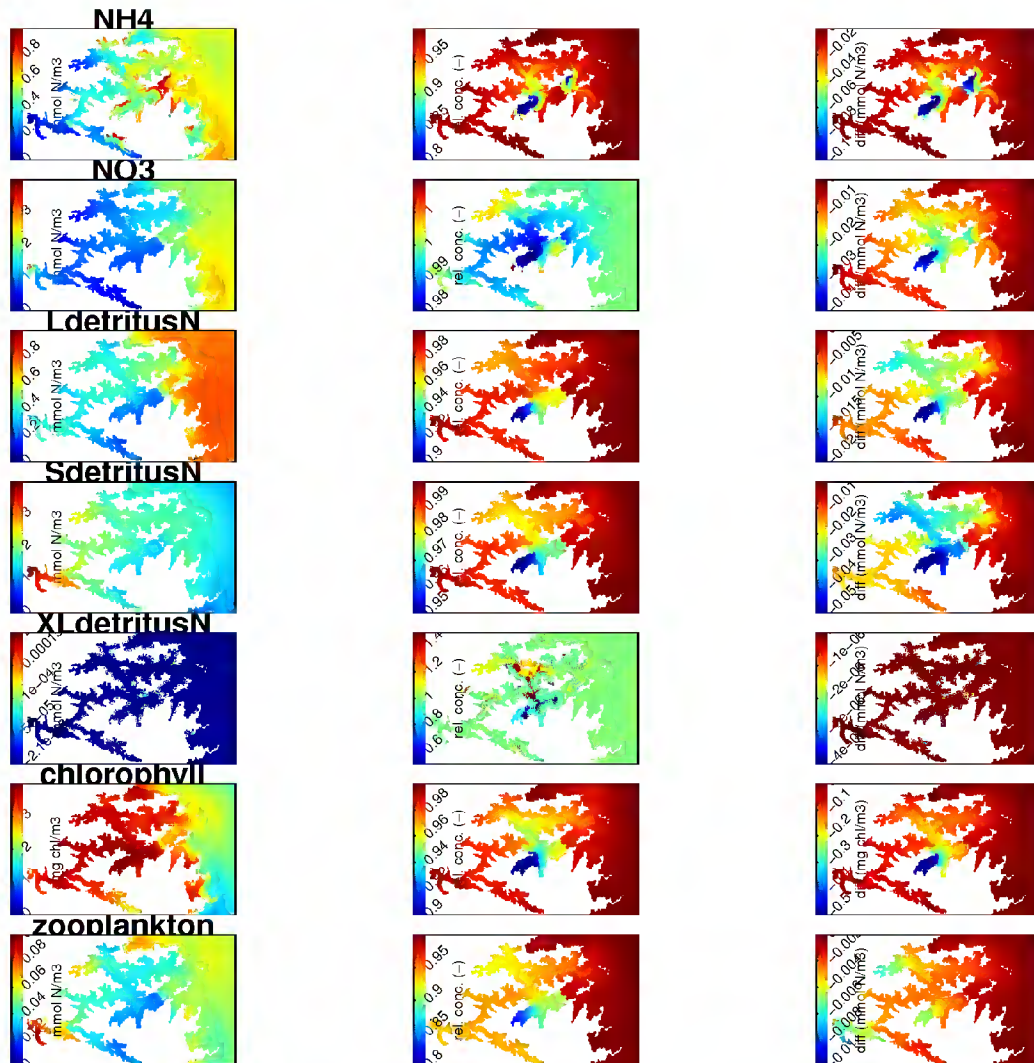


Figure 5-16: Comparison of summer time-averaged surface-layer concentrations in the EM-EF-WD and EM-NF-WD scenarios. Refer to the caption of Figure 5-13 for further explanation. These results are from simulations made with 200 m horizontal resolution.

5.4.3 “Existing mussel/existing fish/with denitrification” versus “no mussel/no fish/with denitrification”

Figure 5-17 (near-surface, winter) and Figure 5-18 (near-surface, summer) illustrate the differences between the EM-EF-WD scenario and the NM-NF-WD scenario. As one might anticipate (on the basis of the results presented in preceding subsections), the effects of removing the mussel farms (lower concentrations of dissolved nutrient, higher concentrations of detrital nitrogen and zooplankton, but mixed increases and decreases of chlorophyll) dominate over the effects of removing the existing fish farms during the winter. The differences between EM-EF-WD and NM-NF-WD tend to be greater in summer than winter.

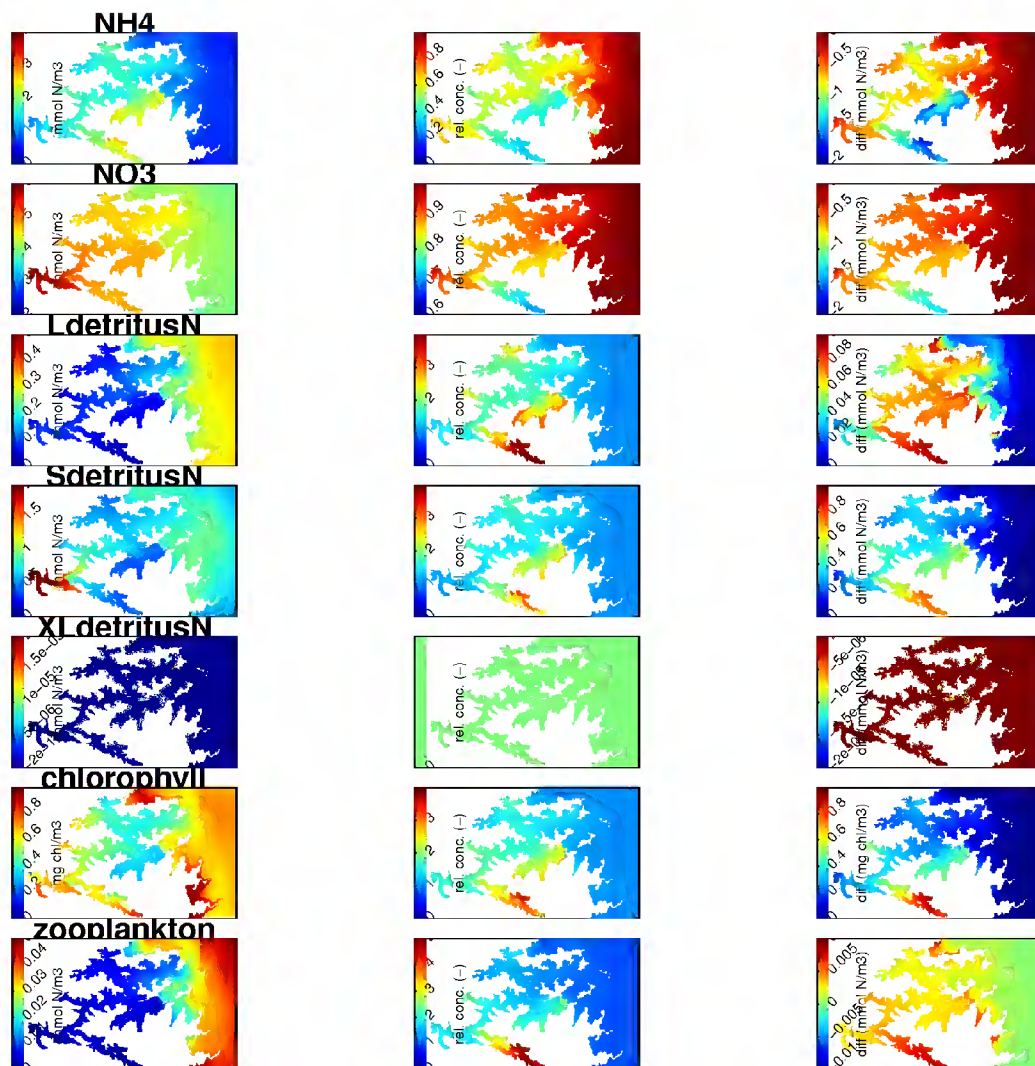


Figure 5-17: Comparison of winter time-averaged surface-layer concentrations in the EM-EF-WD and NM-NF-WD scenarios. Refer to the caption of Figure 5-13 for further explanation. These results are from simulations made with 200 m horizontal resolution.

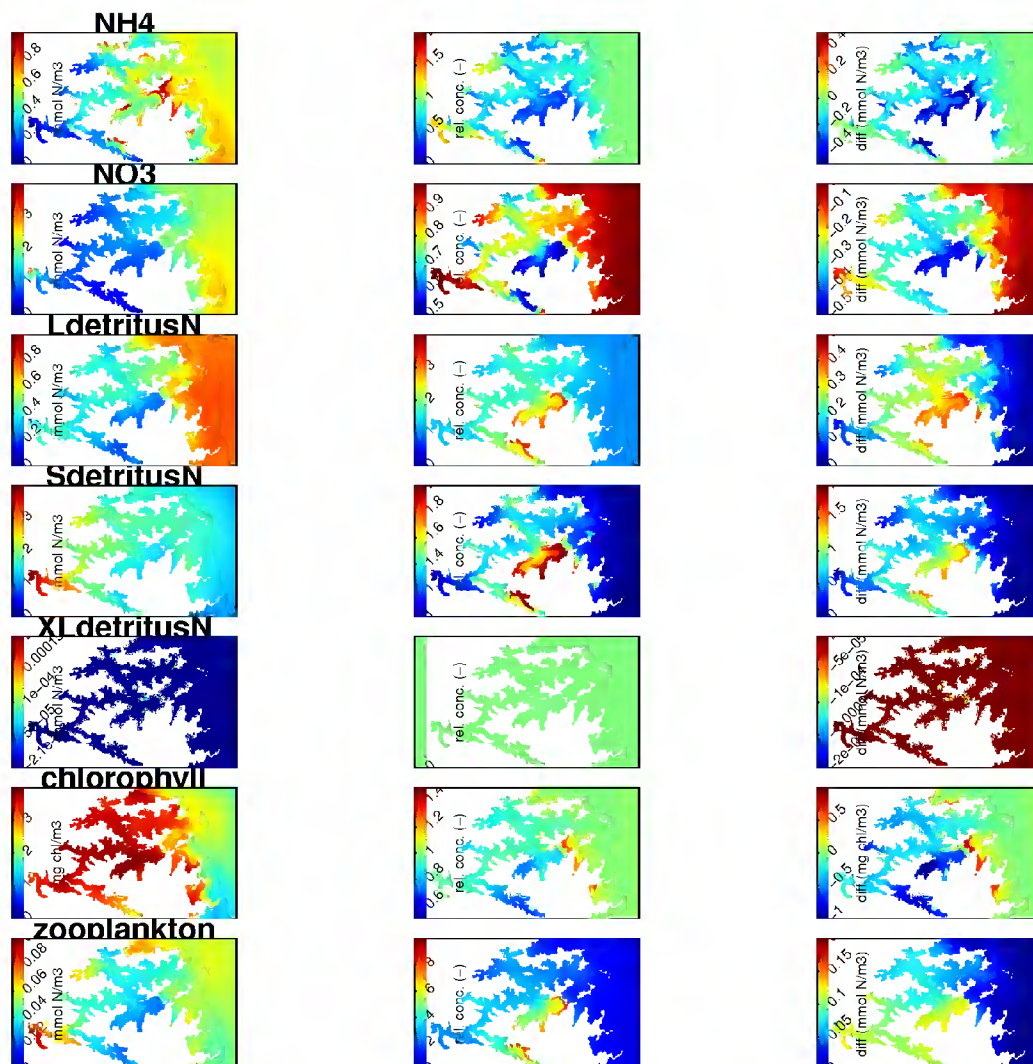


Figure 5-18: Comparison of summer time-averaged surface-layer concentrations in the EM-EF-WD and NM-NF-WD scenarios. Refer to the caption of Figure 5-13 for further explanation. These results are from simulations made with 200 m horizontal resolution.

5.4.4 “Existing mussel/existing fish/with denitrification” versus “approved mussel/approved fish/with denitrification”

Figure 5-19 (surface layer, winter) and Figure 5-20 (surface-layer, summer) illustrate the differences between the EM-EF-WD and AM-AF-WD scenarios.

During winter, ammonium concentrations in outer Pelorus are predicted to rise by 10-20% (driven by the presence of the new salmon farms at Waitata, Richmond and Port Ligar). There is also a small increase within Beatrix/Craill/Clova (driven by the new fish farm within Beatrix Bay). There are some very large changes in the relative concentration of XLdetritus – but these arise in regions where the background concentration is tiny. The absolute changes in the concentration of XLdetritus are very, very small ($<1 \mu\text{mol N m}^{-3}$) in comparison with total (SdetritusN+LdetritusN+XLdetritusN) detrital N concentrations (\sim tens of mmol N m^{-3}). Changes in the concentrations of other state-variables

(LdetritusN, SdetritusN, phytoplankton N and chlorophyll, and zooplankton) are small (up to 5% for zooplankton in Kenepuru, but usually <2% for zooplankton elsewhere and for other state-variables).

During summer, the AM-AF-WD yields higher concentrations of all state-variables throughout most of Pelorus. Ammonium concentrations show the biggest relative changes (more than doubling in the immediate vicinity of the new fish farms). Nonetheless, even close to the fish-farms, the ammonium concentrations remain well below those considered toxic to marine organisms (Anon 2000).

Phytoplankton (as chlorophyll) concentrations are predicted to rise by up to about 10% (less than $0.5 \text{ mg Chl m}^{-3}$) in the vicinities of Beatrix Bay and Waihinau/Port Ligar (near the Waitata and Port Ligar farms). They are predicted to rise by up to $0.2 \text{ mg Chl m}^{-3}$ throughout the remainder of outer and central Pelorus, Tawhitinui Reach and Crail/Clova Bay. Zooplankton concentrations are predicted to increase throughout all parts of Pelorus Sound. The biggest increases are at the head of Pelorus, Kenepuru, Beatrix/Crail/Clova and head of Tawhitinui. In these regions, the increase is around 10–15%.

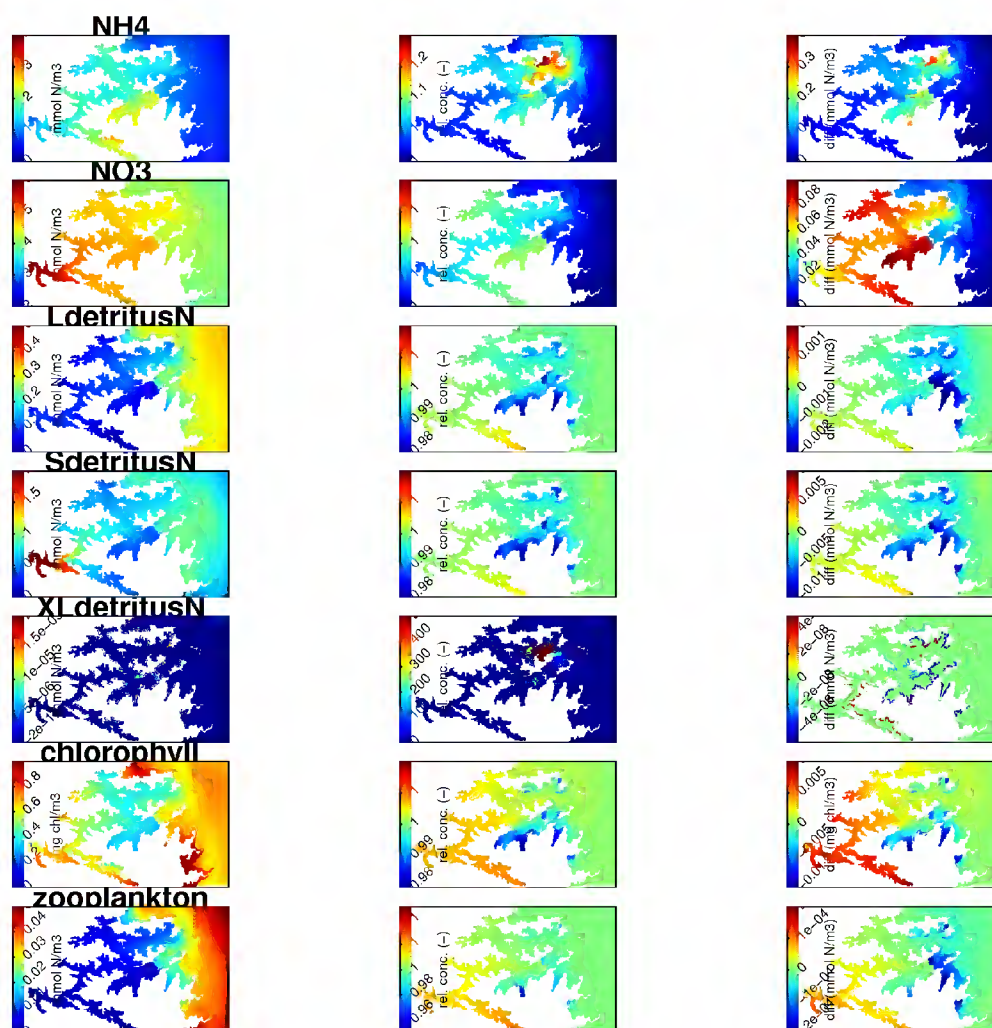


Figure 5-19: Comparison of winter time-averaged surface-layer concentrations in the EM-EF-WD and AM-AF-WD scenarios. Refer to the caption of Figure 5-13 for further explanation. These results are from simulations made with 200 m horizontal resolution.

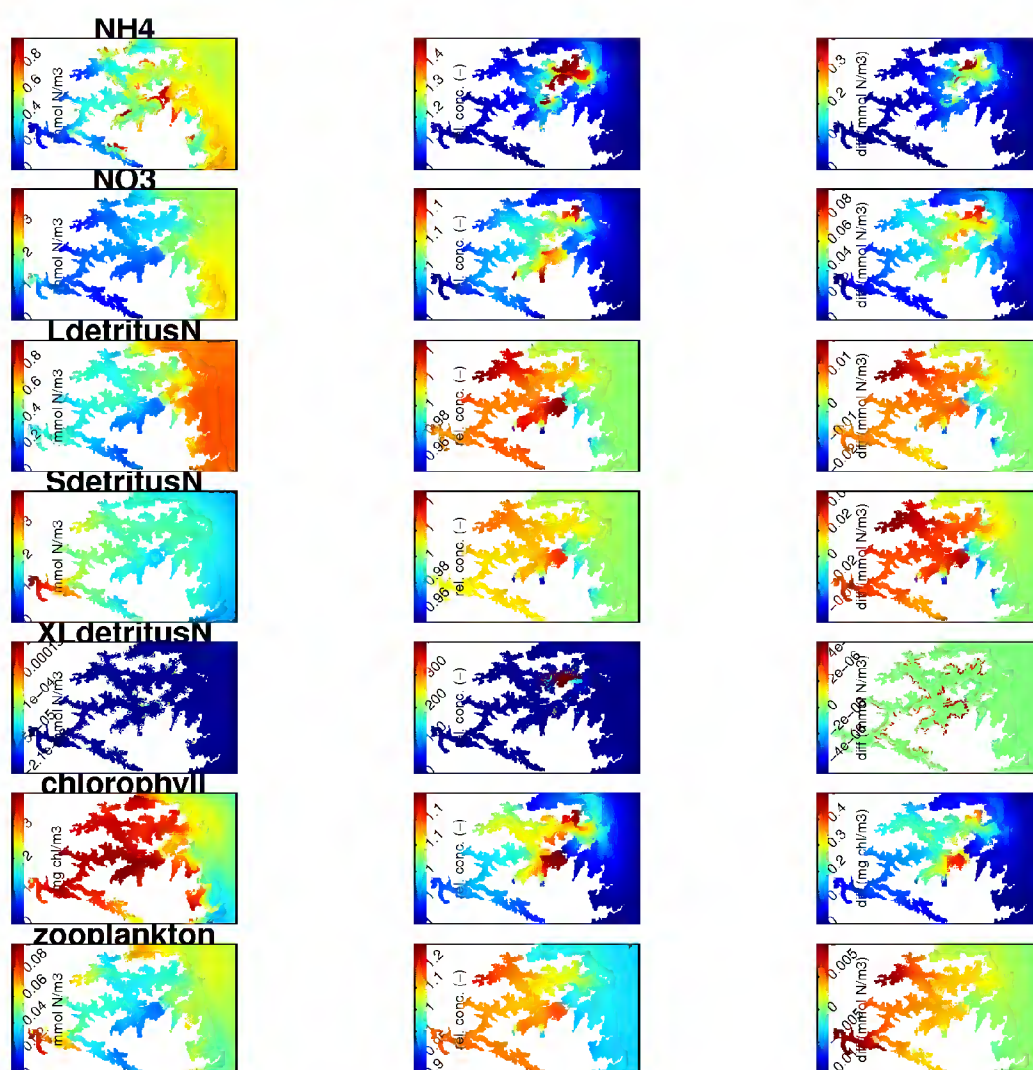


Figure 5-20: Comparison of summer time-averaged surface-layer concentrations in the EM-EF-WD and AM-AF-WD scenarios. Refer to the caption of Figure 5-13 for further explanation. These results are from simulations made with 200 m horizontal resolution.

5.4.5 “Existing mussel/no fish/with denitrification” versus “existing mussel/no fish/no denitrification”

Figure 5-21 (winter, surface layer) and Figure 5-22 (summer, surface layer) illustrate the differences between the EM-NF-WD³⁴ and EM-NF-ND results. As expected, when benthic denitrification is turned off, the system retains more nitrogen – and this effect is greatest in the shallower regions that are far from the Cook Strait mouth of Pelorus.

During winter, ammonium concentrations are predicted to rise by up to about 50% within Hikapu & Popoure reaches and Kenepuru. Nitrate is also predicted to rise by up to about 25% in inner Kenepuru (and 10–20% elsewhere). Phytoplankton concentrations increase very slightly (max

³⁴ Note, this reference scenario (EM-NF-WD) differs from the one (EM-EF-WD) that has been adopted previously.

increase circa $0.01 \text{ mg Chl m}^{-3}$ within Kenepuru). Zooplankton concentrations are predicted to almost double in inner Kenepuru and to increase by 10–50% elsewhere in inner Pelorus.

During the summer, the changes are of similar nature, but larger magnitude. For example, chlorophyll concentrations are predicted to rise almost two-fold (an extra $3\text{--}4 \text{ mg Chl m}^{-3}$) in inner Kenepuru and zooplankton concentrations are predicted to rise more than 10-fold in that region.

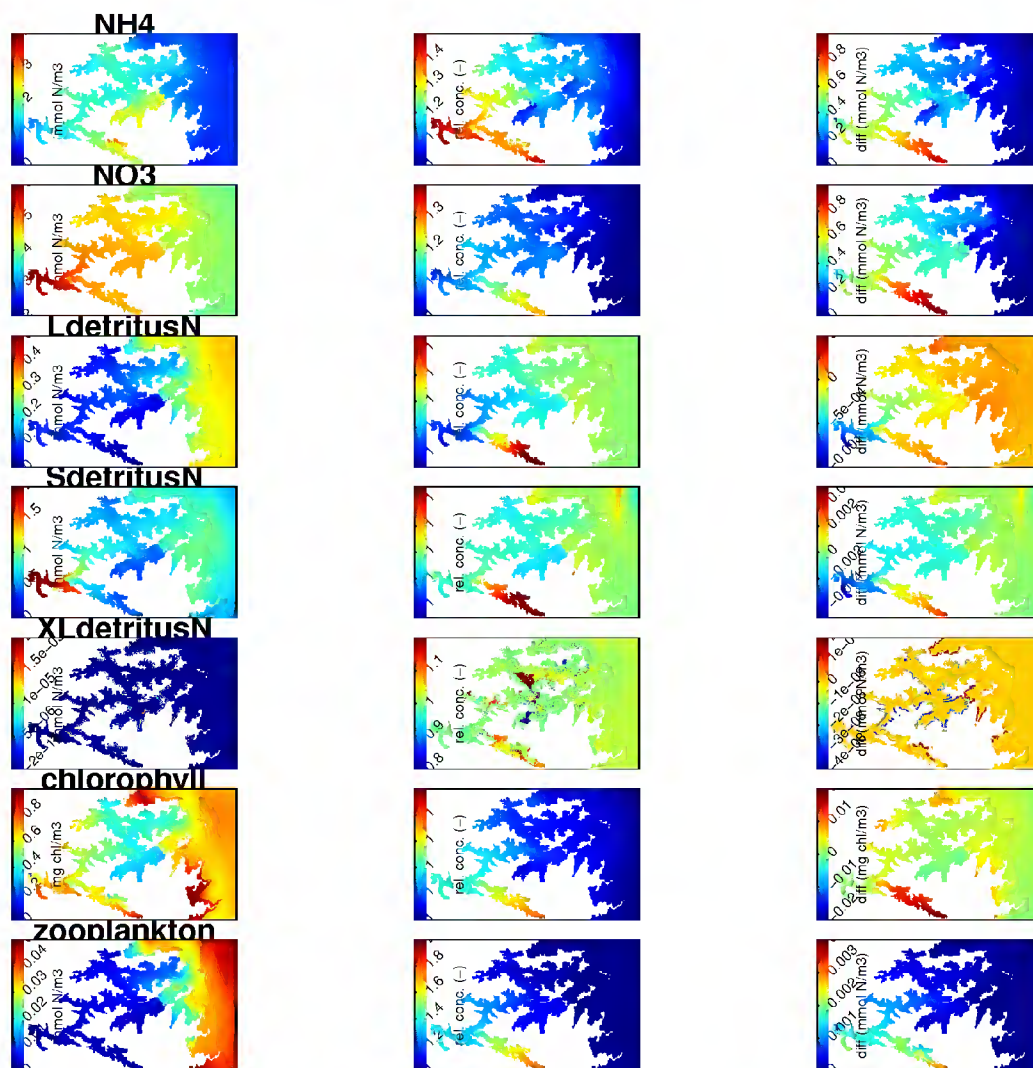


Figure 5-21: Comparison of winter time-averaged surface-layer concentrations in the EM-NF-WD and EM-NF-ND scenarios. Note that the reference simulation is EM-NF-WD rather than EM-EF-WD that was used for most other plots. Refer to the caption of Figure 5-13 for further explanation. These results are from simulations made with 200 m horizontal resolution.

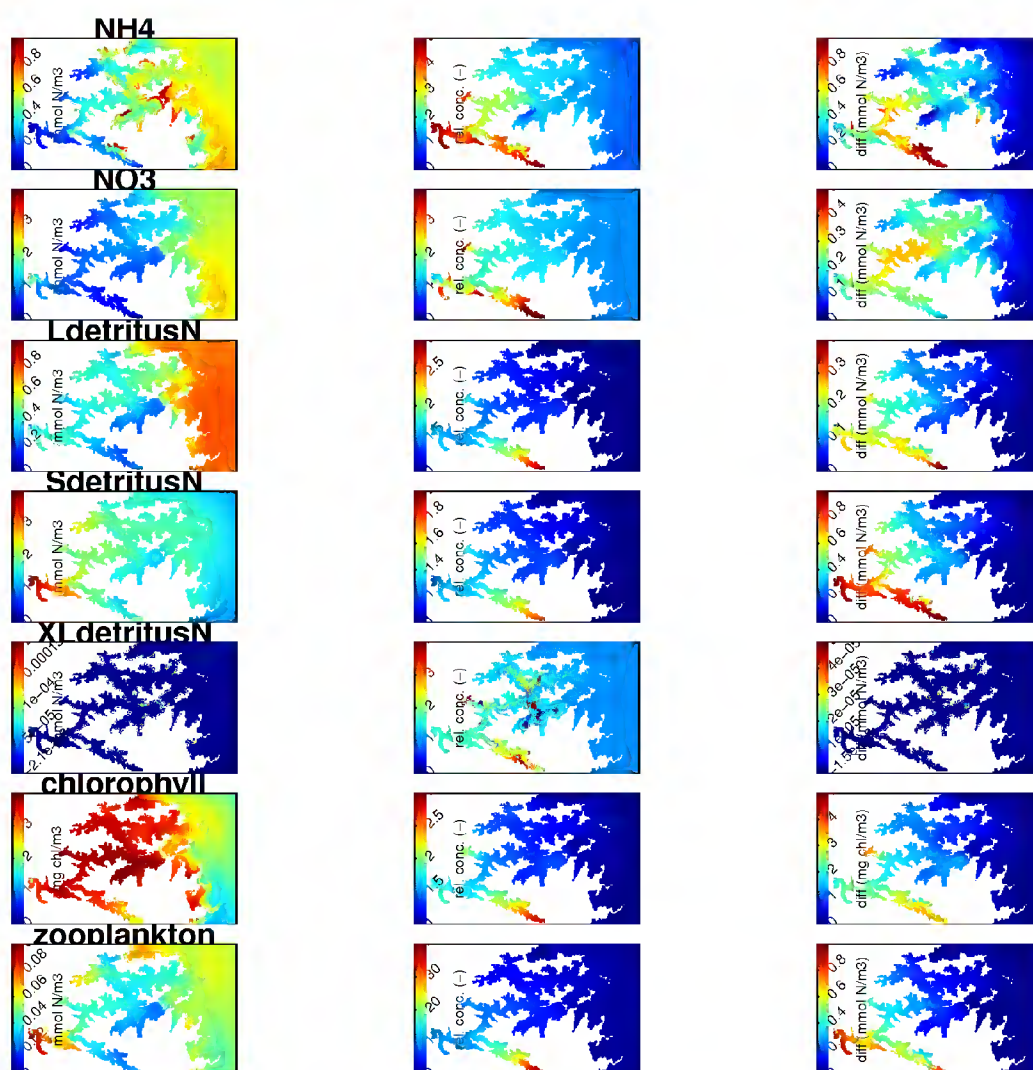


Figure 5-22: Comparison of summer time-averaged surface-layer concentrations in the EM-NF-WD and EM-NF-ND scenarios. Note that the reference simulation is EM-NF-WD rather than EM-EF-WD that was used for most other plots. Refer to the caption of Figure 5-13 for further explanation. These results are from simulations made with 200 m horizontal resolution.

5.4.6 “Existing mussel/existing fish/with denitrification” versus “approved mussel/approved fish/no denitrification”

Finally, in Figure 5-23 (winter, surface layer) and Figure 5-24 (summer, surface layer), results from the EM-EF-WD and AM-AF-ND simulation are compared. As expected, the combination of turning off nitrogen removal and adding more nitrogen into the system (from the additional fish farms) causes the concentrations of all state-variables to rise (the few additional mussel farms cannot induce sufficient depletion to offset this). Again, the changes tend to be much greater in summer than winter. Even in the summer, however, ammonium concentrations remain low relative to those that are toxic and time-averaged chlorophyll concentrations remain below 5 mg Chl m^{-3} throughout the Sounds³⁵.

³⁵ In the EPA Board of Inquiry decision into New Zealand King Salmon Ltd’s application for additional farms, chlorophyll concentrations that are persistently (annual average) greater than $5 \text{ mg Chl a m}^{-3}$ were deemed to be unacceptable Whiting, G., Beaumont, H., Ellison, E., Farnsworth, M., Briggs, M. (2012) Board of Inquiry New Zealand King Salmon requests for plan changes and applications for resource consents: 356.

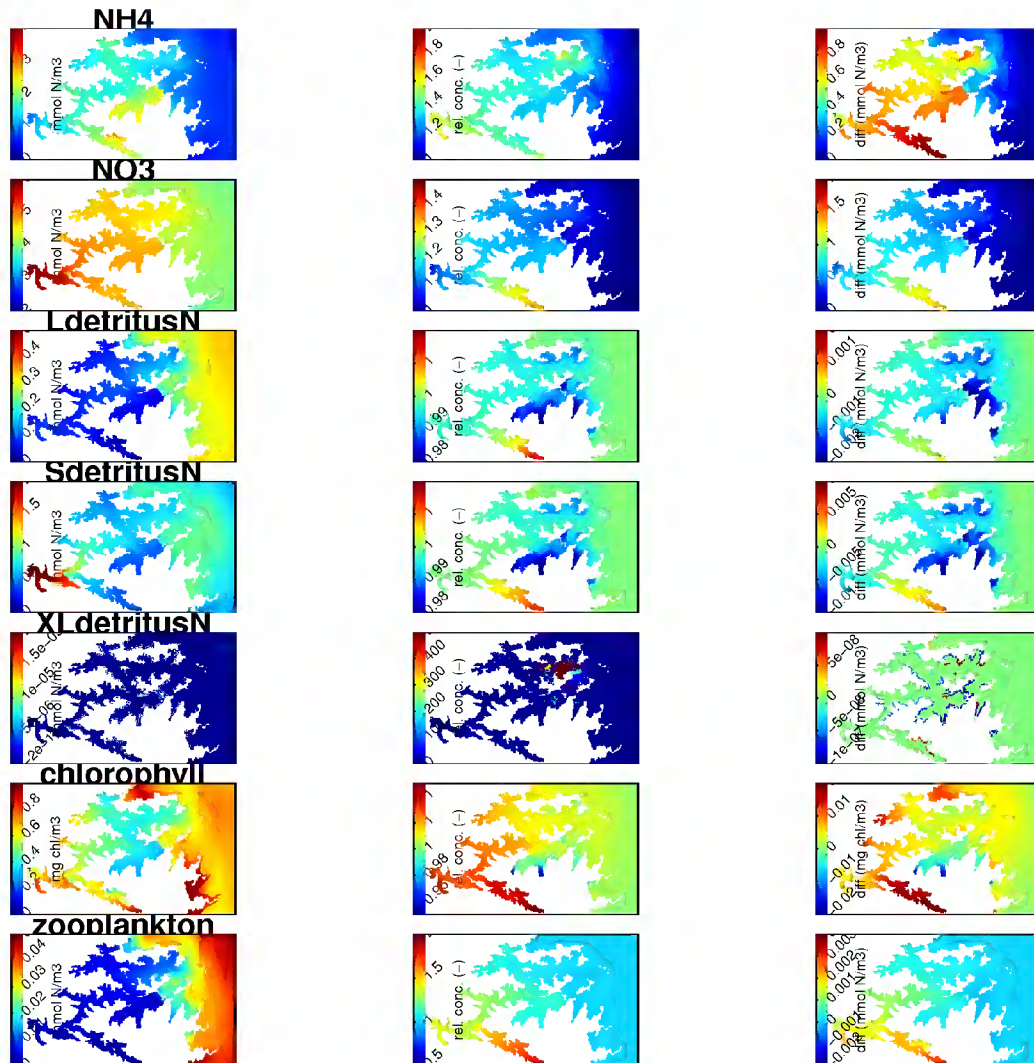


Figure 5-23: Comparison of winter time-averaged surface-layer concentrations in the EM-EF-WD and AM-AF-ND scenarios. Refer to the caption of Figure 5-13 for further explanation. These results are from simulations made with 200 m horizontal resolution.

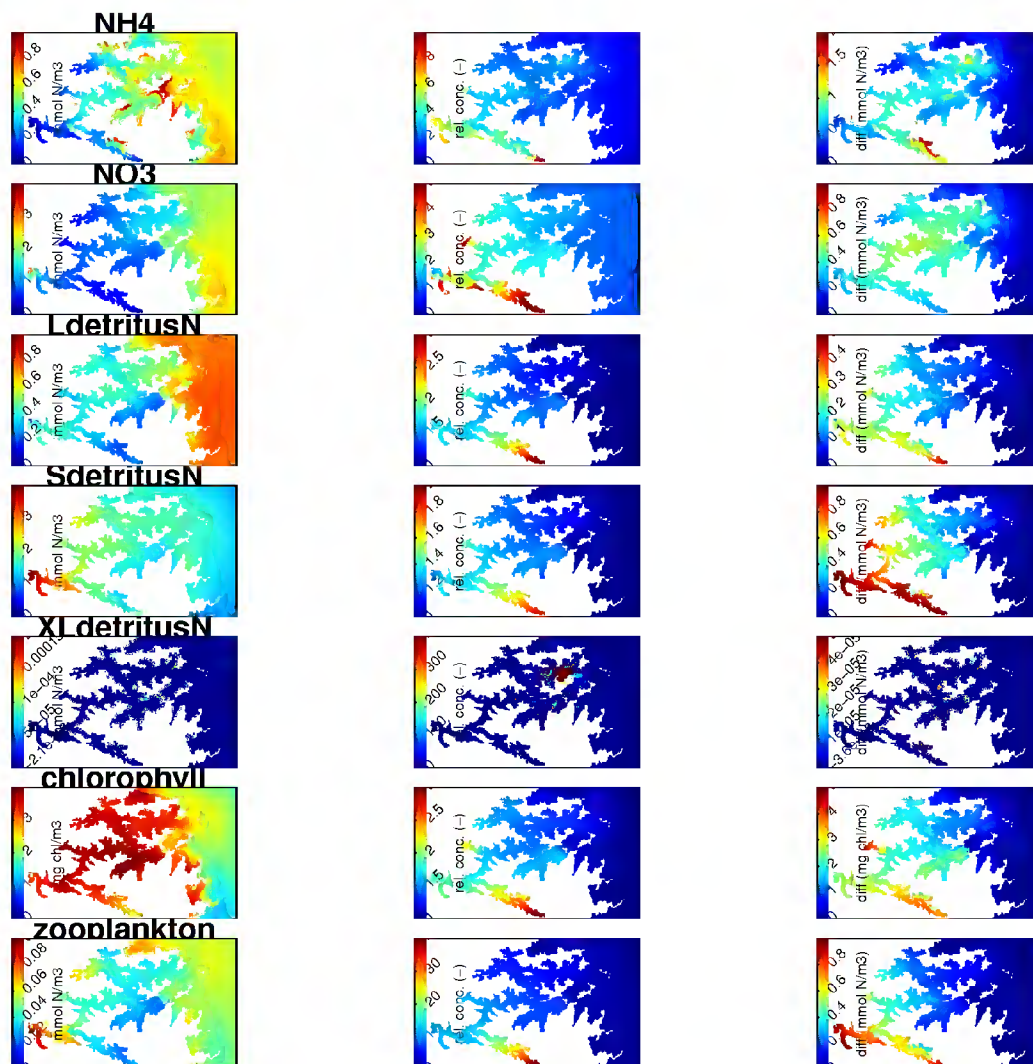


Figure 5-24: Comparison of summer time-averaged surface-layer concentrations in the EM-EF-WD and AM-AF-ND scenarios. Refer to the caption of Figure 5-13 for further explanation. These results are from simulations made with 200 m horizontal resolution.

5.5 Concentration changes in near-bed waters

The preceding sections have shown results from the surface-most layer of the model. We believe that the patterns will be similar throughout the vertical extent of the surface mixed layer that exists for much of the year within the Pelorus system. There are, however, reasons to believe that patterns may differ below the surface mixed layer. Firstly, there is less light at depth, so phytoplankton will be less able to respond to any fish-farm derived nutrient. Secondly, the mussel and fish-farms will not extend far (if at all) below the mixed layer, so seston will feel only weak (or indirect) effects of the farms. Finally, the main channel of Pelorus exhibits a strong estuarine flow. Water in the upper 20 m or so tends to flow out of the Sound into Cook Strait whilst deeper water flows into the Sound from Cook Strait.

Appendix E presents pictures (of the near-bed layer) that are conceptually equivalent to those presented for the surface layer within the preceding sub-sections. As one might expect:

- “Cook Strait-like” water-properties extend far further into Pelorus near-bed than near-surface. Furthermore, the change from Cook Strait-like to (inner) Pelorus-like is more gradual at depth (c.f. almost a sharp front at the Cook Strait mouth in the near-surface layer).
- Nutrient concentrations tend to be higher near-bed than near-surface (esp. in summer). Conversely, plankton concentrations tend to be lower.
- Effects of mussel farms and fish farms upon seston tend to be smaller near-bed than near-surface.
- In some instances, the effects of mussels and fish-farms upon nutrient may be greater near-bed (presumably, because XLdetritus from the farms quickly sinks to the bed and a fraction of that mineralizes into nutrient that returns the bottom-most layer).
- Turning off benthic denitrification has a larger impact upon near-bed nutrient concentrations than it has upon near-surface concentrations.

Despite these differences, the general inferences that can be drawn from the near-bed plots are similar to those that can be drawn from the near-surface ones:

- Mussel-farming (at today’s extent) has greater influence upon water-quality than fish-farming (at today’s extent).
- Increasing the scale of mussel and fish farming will induce further change (almost universally amounting to <10% percent of today’s conditions (in a time-average), depending upon which property one chooses to examine and when/where one chooses to look).
- In the unlikely event that all benthic denitrification within the Pelorus system were to cease and mussel and fish-farming were to expand from today’s operational farms to all consented farms, the time-averaged chlorophyll and detritus concentrations may double in the central and inner (especially Kenepuru) regions, but the seasonal average would remain below 5 mg Chl m⁻³.
- The water-quality of Kenepuru and Beatrix/Crail/Clova (and, to a lesser extent, the landward reaches of Tawhitinui Reach) are more sensitive to changes in denitrification, mussel-farming and (even spatially distant) fish-farming than are the parts of the Sound that lie close to (or within) the main axis of estuarine flow.

5.5.1 Simulated changes of instantaneous water-quality at the seven MDC sampling stations

Figure 5-25 to Figure 5-31 illustrate the simulated dynamics of each state-variable in the near-surface layer at each MDC water-quality sampling station under five of the six different scenarios³⁶. As noted earlier, the primary purpose of these plots is to demonstrate that the time-averaging process that was employed to generate the false colour plots (presented in the previous section) has not masked short-lived-but-much-larger differences between the dynamics stemming from differing scenarios. The plots certainly reveal that the time-averaging masks substantial high frequency (days to fortnight time scale) variations within any one scenario, but the patterns of fluctuations are similar across all scenarios. Thus, within any given season, the magnitudes of differences between scenarios remains similar day-to-day. The time-averaging is not masking short-lived-but-much-larger between scenario differences.

That is not to say that the between scenario differences are always small. Clearly:

- simulated 'present day' dynamics (EM-EF-WD) are markedly different from those of a notional, historical system in which catchment and oceanic inputs remain the same as they are now but neither mussel, nor fish-farms are present (NM-NF-WD),
- similarly, the model simulations indicate that, were benthic denitrification to cease *throughout* the domain (scenarios EM-EF-ND and AM-AF-ND), the system's dynamics would change markedly.

On the other hand, the model simulations indicate that the incremental changes (from the 'existing situation') associated with adding already-approved-but-not-operating marine farms will be small at the seven Marlborough District Council water quality sampling stations³⁷.

³⁶ To avoid generating even more cluttered/confusing plots, the sixth scenario was excluded from these plots. Results from that scenario are no more extreme.

³⁷ The MDC sampling stations are not close to any of the forthcoming new fish- or mussel farms. The incremental changes in the immediate vicinity of the new farms may be larger (see for example, Figure 5-19 & Figure 5-20).

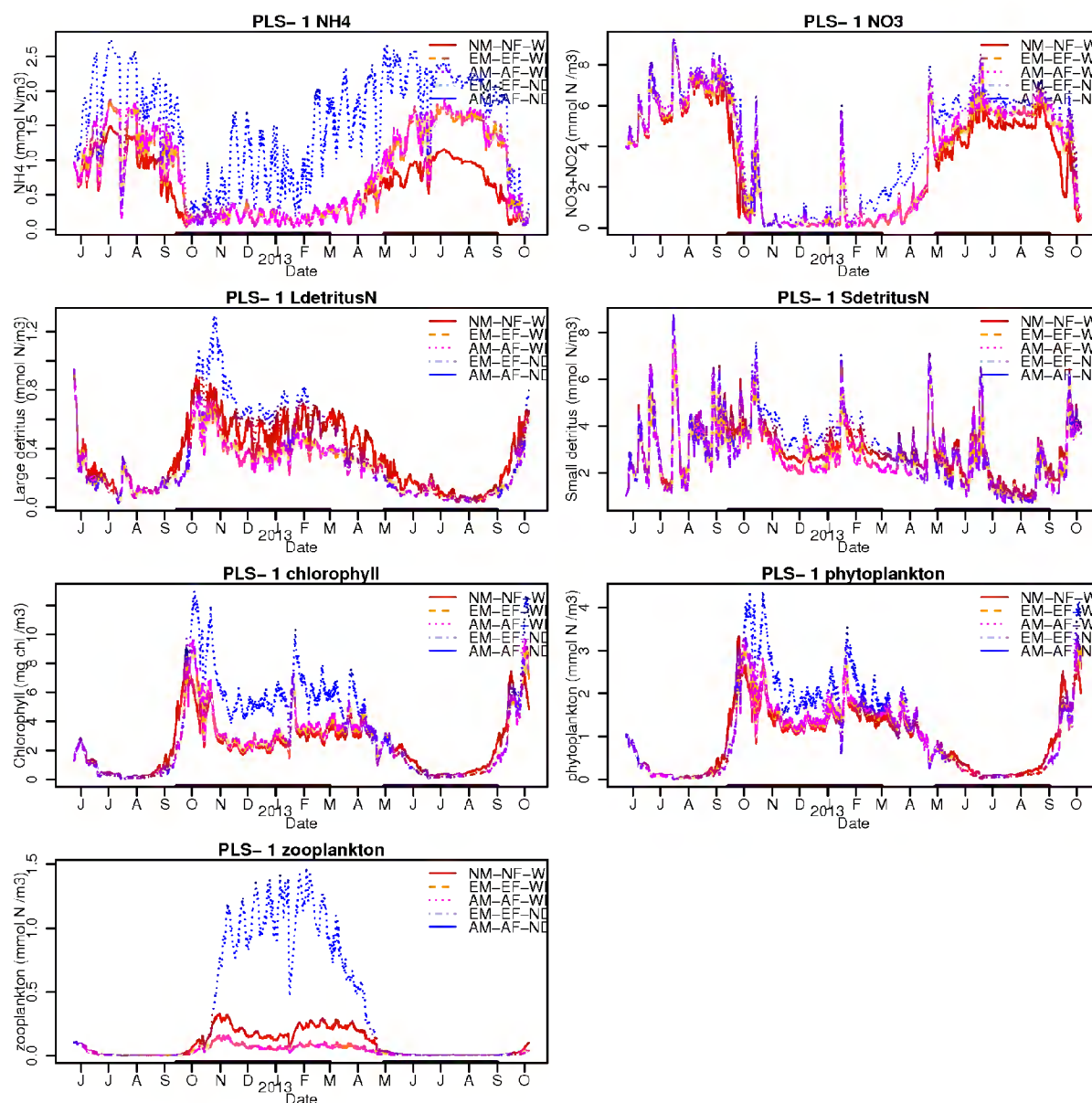


Figure 5-25: Simulated time-series of each state-variable in the surface-most layer at station 1 for five scenarios. The thick black lines running along the bottom of each plot indicate the periods used to produce the time-averaged plots. In this figure and the subsequent ones (for other MDC sites), the results for the EM-EF-ND and AM-AF-ND scenarios are almost coincident with one another (such that it is difficult to distinguish two separate lines). Similarly, the results from EM-EF-WD and AM-AF-WD are almost indistinguishable.

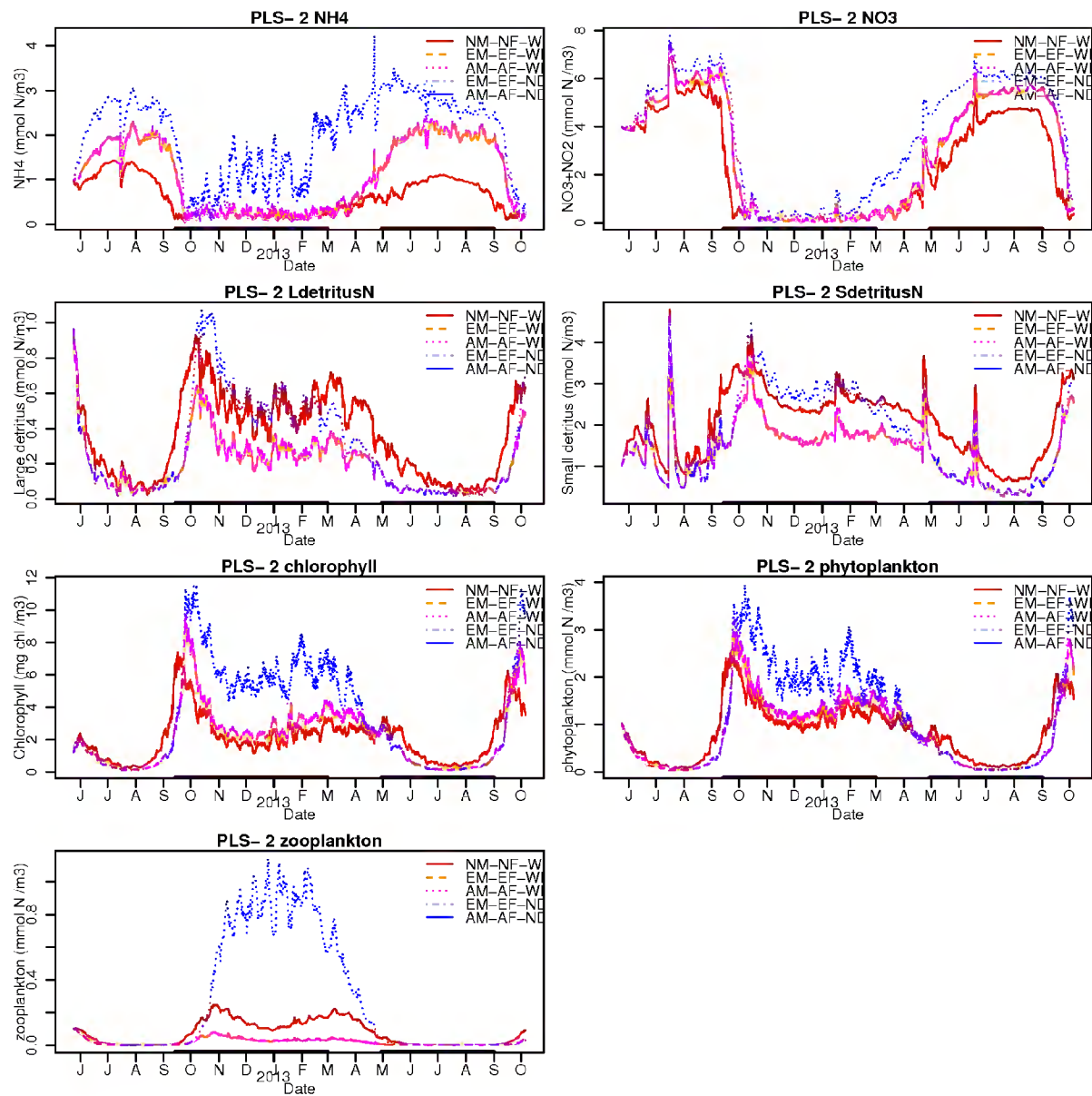


Figure 5-26: Simulated time-series of each state-variable in the surface-most layer at station 2 for five scenarios.

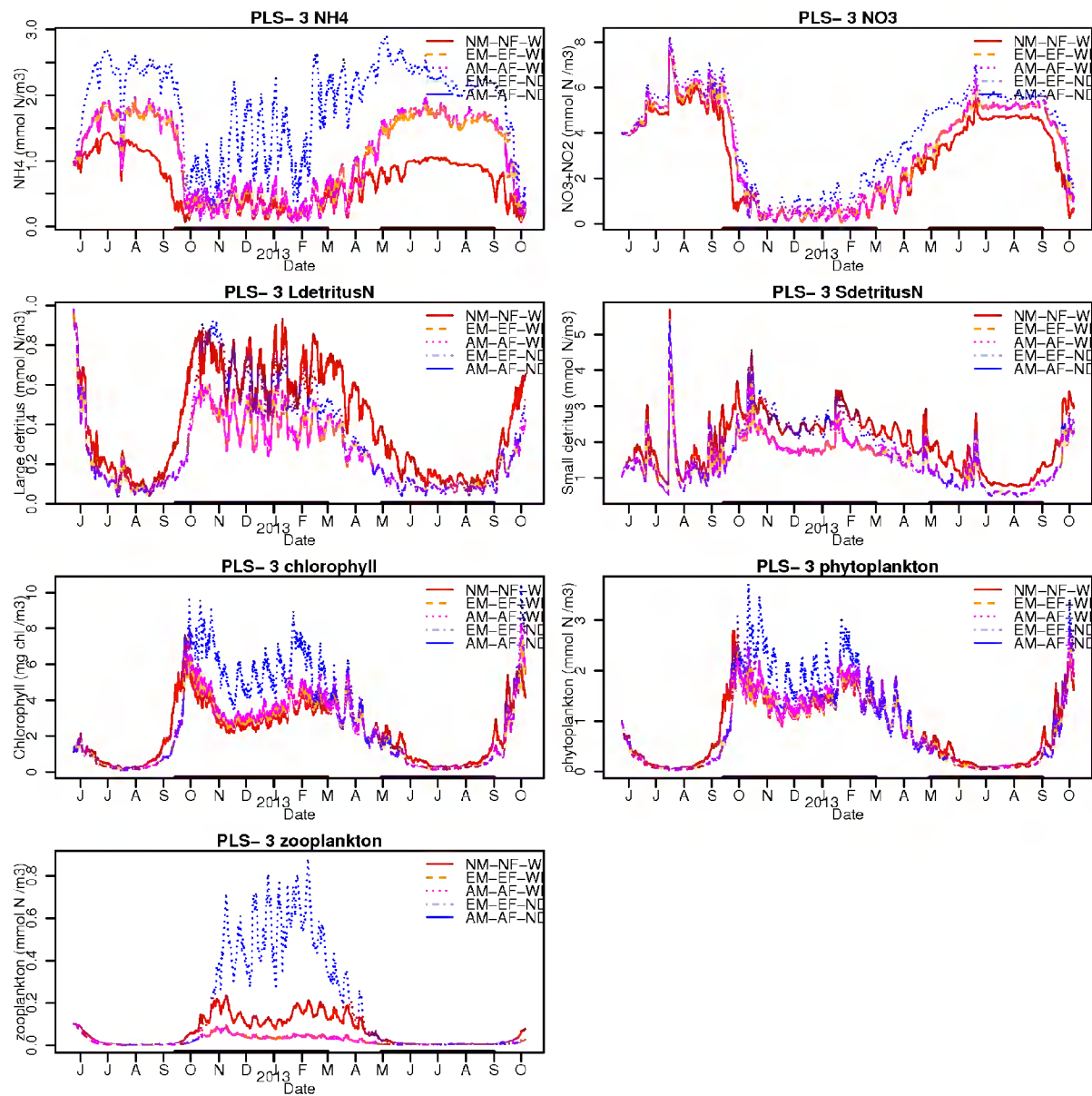


Figure 5-27: Simulated time-series of each state-variable in the surface-most layer at station 3 for five scenarios.

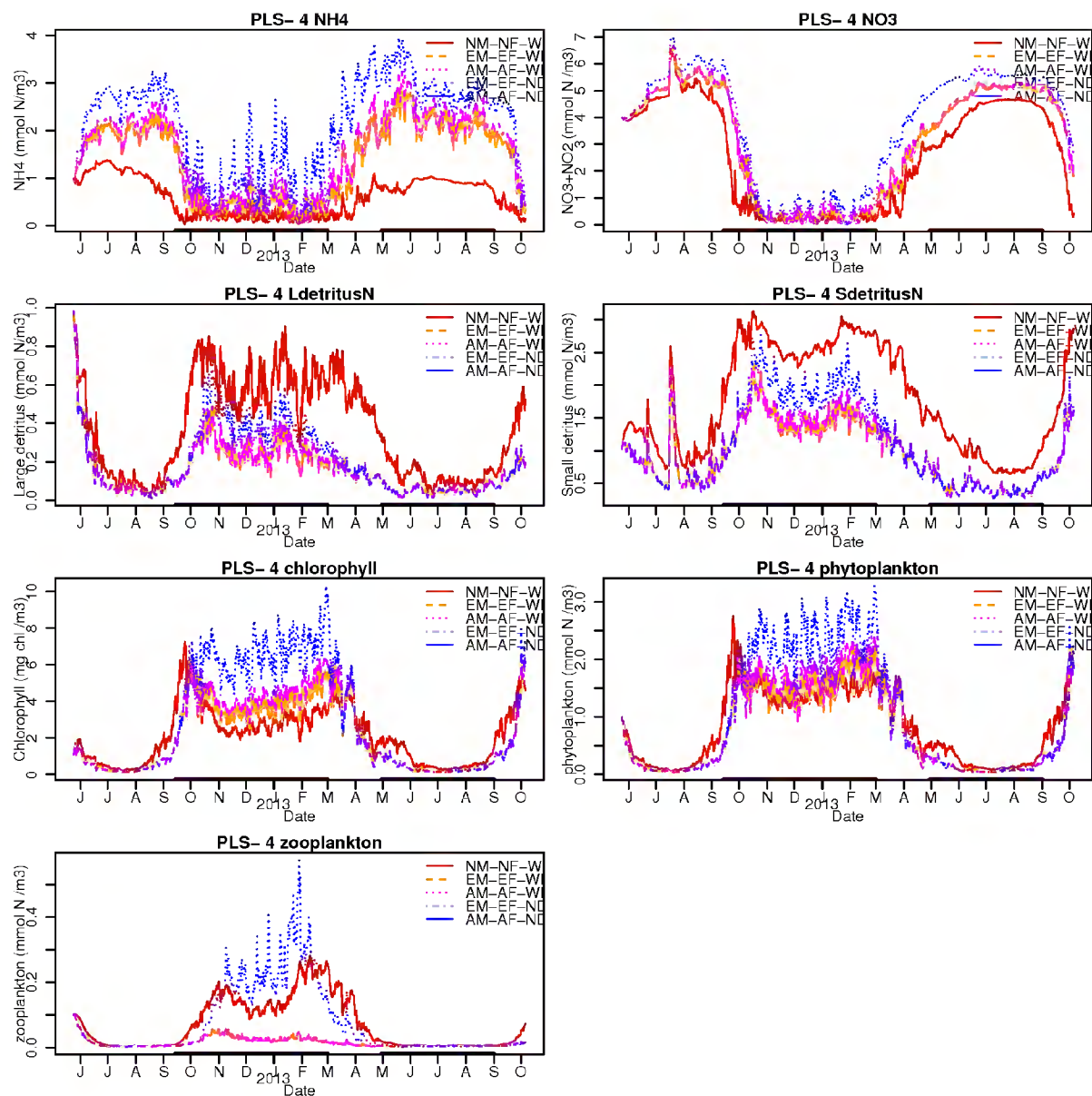


Figure 5-28: Simulated time-series of each state-variable in the surface-most layer at station 4 for five scenarios.

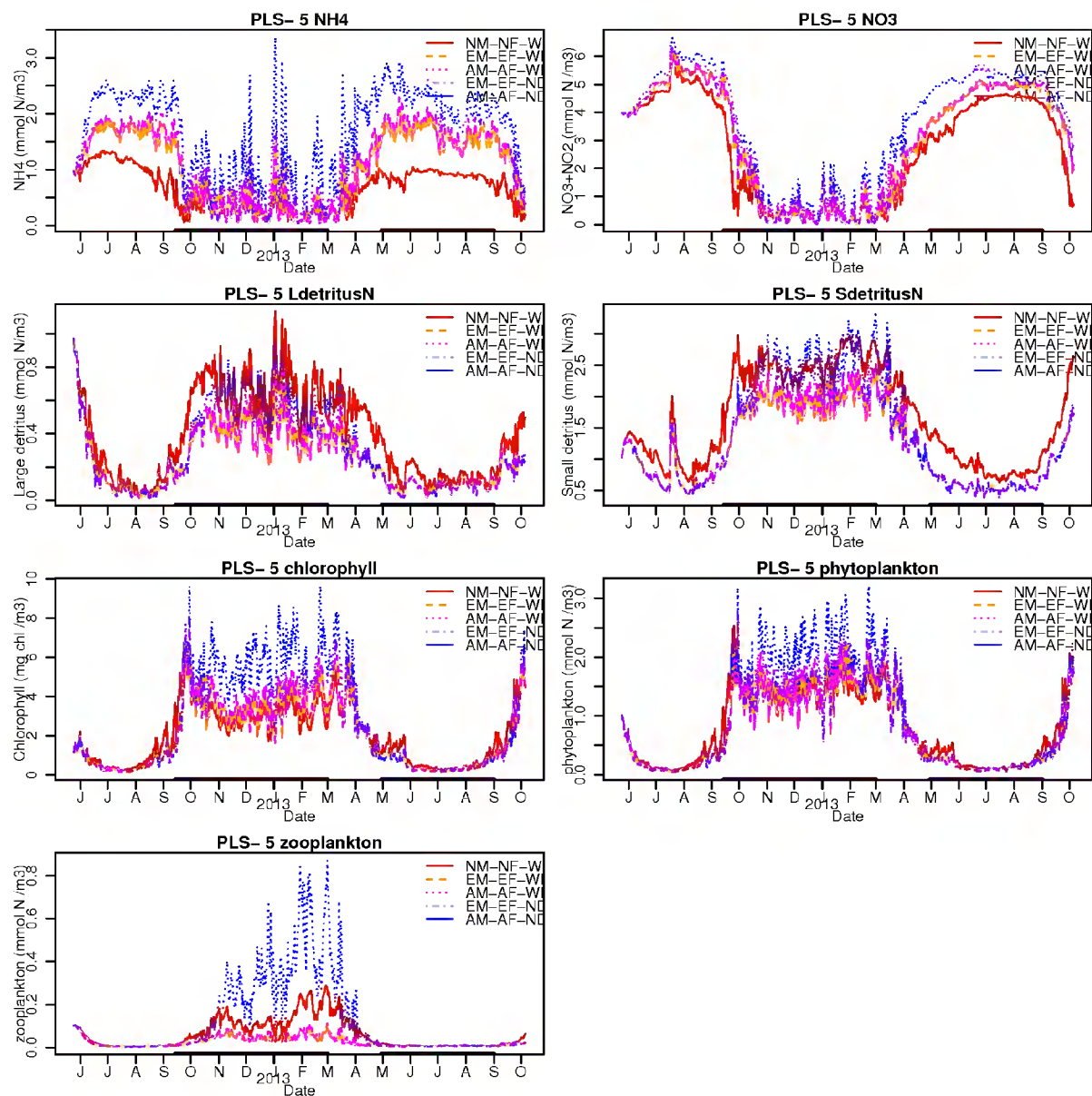


Figure 5-29: Simulated time-series of each state-variable in the surface-most layer at station 5 for five scenarios.

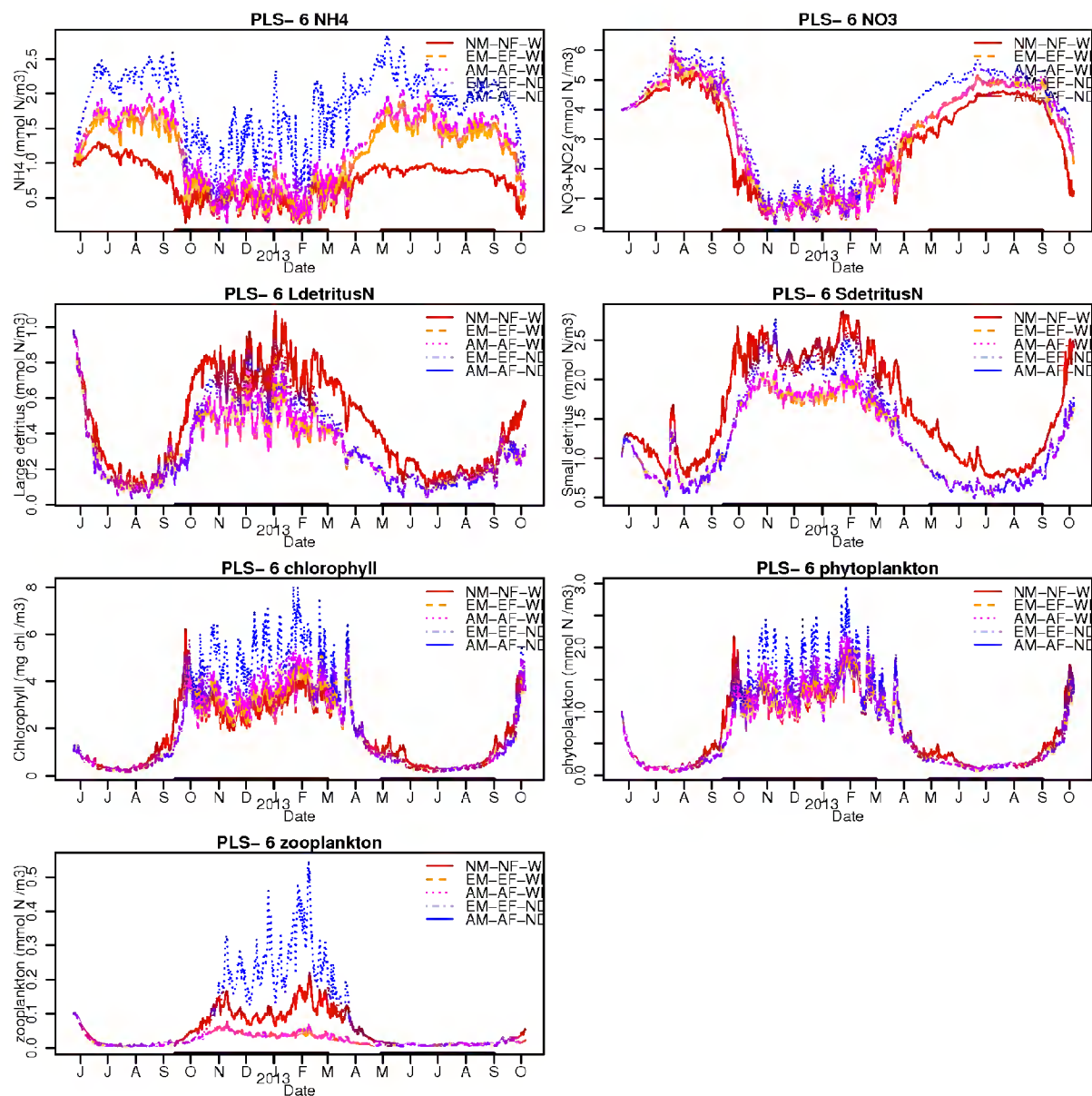


Figure 5-30: Simulated time-series of each state-variable in the surface-most layer at station 6 for five scenarios.

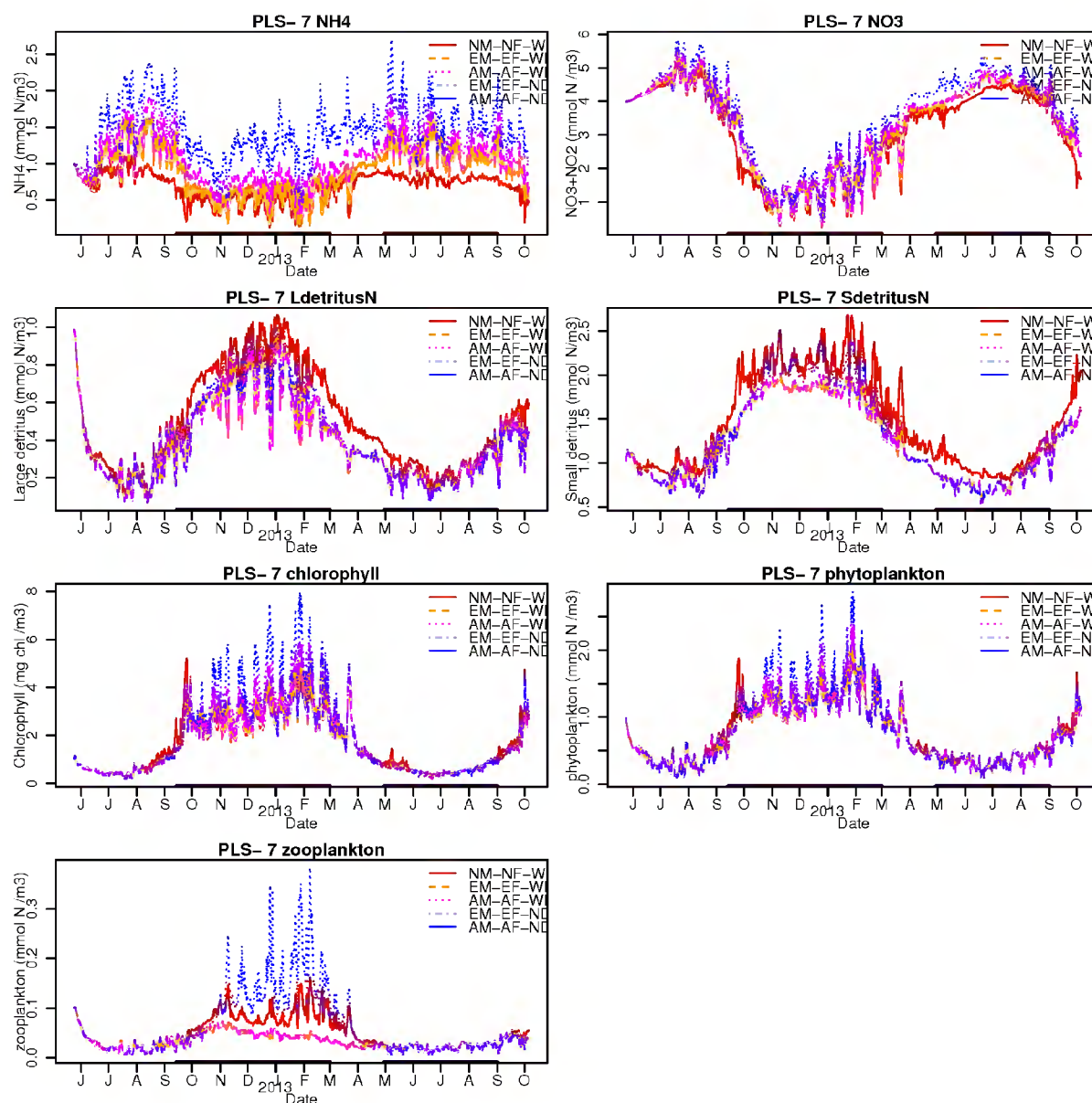


Figure 5-31: Simulated time-series of each state-variable in the surface-most layer at station 7 for five scenarios.

5.6 Comparison of the farming induced nitrogen fluxes

Figure 5-32 illustrates the magnitudes of the mussel farm nitrogen uptake and release fluxes at the whole-of-domain spatial-scale. In the graph, negative values imply that the material in question is suffering net removal from the water-column through the activity of the mussels. Conversely, positive values indicate net addition. The graph reveals that (in the model) the mussels gain the majority of their nutrition from small detrital material (grey) and from phytoplankton (red). This is a simple consequence of the greater abundance of these materials relative to zooplankton and large detritus (which the mussels also consume). The combination of higher summertime water temperatures (permitting greater specific filtration rates by the mussels) and higher summertime seston concentrations imply that the rates of ingestion and faeces/pseudofaeces production are

several times greater in summer than in winter. Ammonium release rates are also higher in summer, but the amplitude of the annual cycle of ammonium production is much smaller than that of (pseudo)faecal production. At the height of summer, the rate of nitrogen release as (pseudo)faeces slightly exceeds the rate of ammonium release, but for most of the year ammonium release rate is substantially greater than rate of release of nitrogenous detritus.

Figure 5-33 illustrates the temporal patterns of nitrogen release as (faeces, XLdetritusN) and as ammonium for the existing fish farms in Pelorus Sound. Ammonium excretion tends to be highest in summer reflecting the influence of water temperature upon basal respiration, and the fact that fish are growing most rapidly at that time of year. It also shows the feed input (as nitrogen). The feed input changes month-to-month in a stepwise manner – reflecting the fact that we made no attempt to smooth (interpolate) daily values from the monthly total feed input rates provided to us by NZKS Ltd. Rather, we calculated an average daily rate for each farm for each month and applied those rates throughout the month. Ammonium and detrital N production rates also change in stepwise manners, but the changes are less abrupt because there are various buffering/smoothing mechanisms within the model. For example, ammonium excretion arising from basal respiration is a significant part of the total ammonium flux. Its production rate is a function of fish size and water temperature but not of feed inputs. Total fish farm nitrogen input into the environment (sum of detritus and ammonium) is circa 1 tonne d⁻¹. During the summer months, this is considerably greater than the net nitrogen release rate from mussel crop (a negative value, indicating net nitrogen removal from the water-column into mussel flesh), but a bit less than the nitrogen loss from the mussel crop during winter (net export of flesh nitrogen into the water-column – indicative of condition loss).

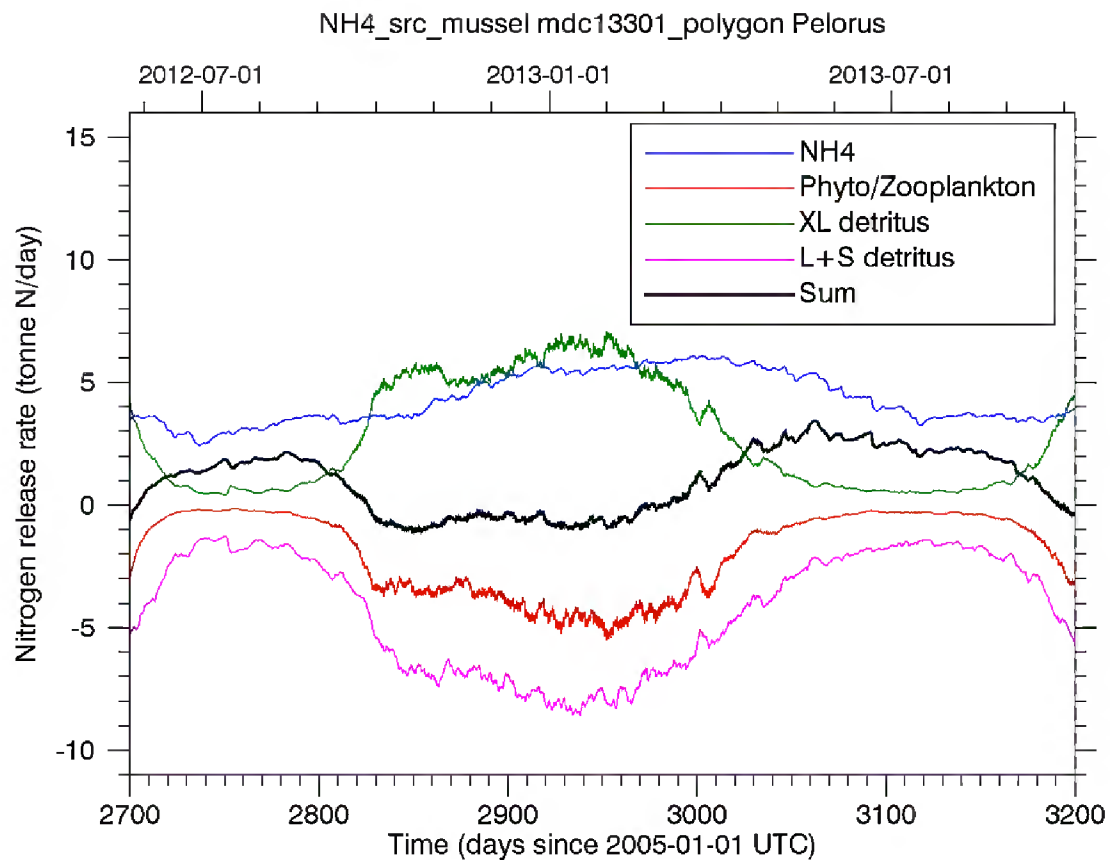


Figure 5-32: Nitrogen uptake (negative) and release (positive) release rates associated with mussel ingestion, respiration and excretion. blue - mussel excretion of ammonium; red – mussel ingestion of nitrogen within plankton; pink – mussel ingestion of small and large detrital nitrogen; green – mussel net egestion of extra large detrital nitrogen (faeces & pseudo-faeces); black – net sum (total egestion - ammonium excretion – total ingestion).

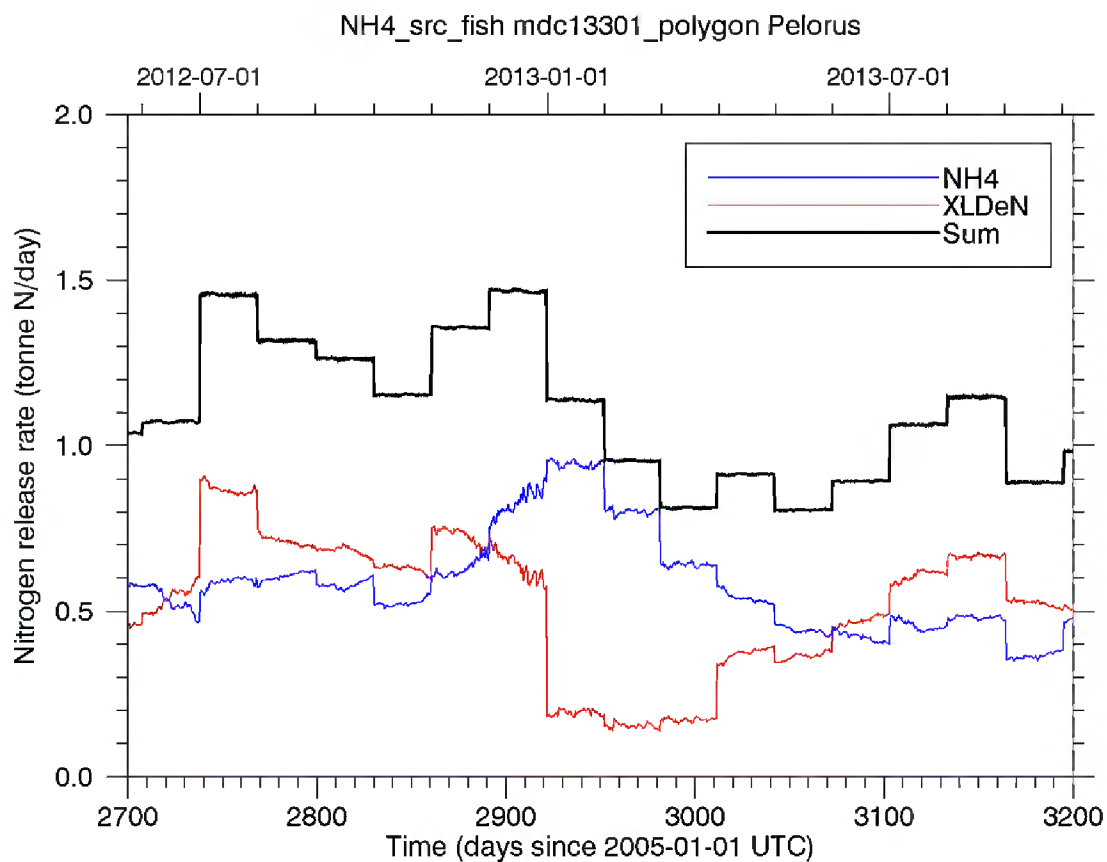


Figure 5-33: Nitrogen release to the water from fish farms. Red: sum of uneaten food and faeces; blue: ammonium excretion; black: sum of excretion and faeces/uneaten food.

6 Biophysical model: Discussion

6.1 Limitations of the biophysical model

Like all models, our biophysical model embodies many simplifications relative to reality. The foodweb is truncated. The highest explicit trophic group is zooplankton. The influence of predators of zooplankton is represented by imposing a specific mortality rate (d^{-1}) upon the zooplankton. In particular, the Fennel model assumes that the specific mortality rate increases linearly with rising zooplankton abundance. This assumption is not atypical of NPZD models, but Steele and Henderson (1992) and Edwards and Yool (2000) have shown that the dynamics of a nutrient-phytoplankton-zooplankton model can be very sensitive to the form (and parameterization) of this top-level predatory closure term. Under some situations, the system can be induced to exhibit high frequency oscillations (alternating booms and busts) even in an otherwise constant environment. The fact that the Fennel model assumes that the specific mortality rate increases linearly with rising zooplankton abundance reduces the likelihood of such oscillations. The implication is that it is possible that the Fennel model may under-estimate the frequency and/or extent of short-lived algal blooms. We have chosen to focus our attention upon time-averages. These are likely to be less sensitive to the form of the mortality closure term.

The model foodweb is deliberately simple. It does not include higher trophic levels. Perhaps more importantly, it does not include bacteria or macroalgae. Like the phytoplankton, bacteria and macroalgae will consume farm-derived nutrients. Since the model lacks these two groups, the phytoplankton have exclusive access to the farm derived nutrient. Since the model phytoplankton do not have to 'share' the fish-farm-derived nutrient with other taxa, it seems probable that the model over-estimates the extent to which the phytoplankton community may change (increase) in response to farm-derived nutrient. Another manifestation of the model's simple foodweb is that it contains only one phytoplankton group. In reality, the phytoplankton community of the Marlborough Sounds is composed of several tens of species. At any instant, phytoplankton biomass will be dominated by only a few species, but the dominant species change in a characteristic way through the year – primarily because differing species have differing nutrient and light requirements. As an emergent property, the apparent (emergent) kinetic 'coefficients' that govern nutrient-uptake and photosynthesis etc., of the real-world phytoplankton population change through the year. Since the model has only one phytoplankton group, it will not mimic these changes well and this may be one of the reasons that the model does not always adequately reproduce all aspects of the field data.

Some phytoplankton taxa are motile – notably dinoflagellates. Dinoflagellates comprise 10-50% of the Pelorus Sound phytoplankton community by biomass (in the $>2\ \mu\text{m}$ fraction). Motile algae have a competitive advantage in stratified waters that have a nutrient-depleted surface layer. This is because motile individuals are able to either: (a) actively hold position at a depth where photosynthetic- and nutrient-acquisition rates can be balanced, or (b) migrate between nutrient-rich deeper waters (where they can replenish their internal nutrient stores) and the light-rich surface layers (where they can replenish their carbon stores). The fact that the phytoplankton of the Fennel model are non-motile may help to explain why it fails to reproduce the deep chlorophyll maxima that are common in some parts of Pelorus Sound.

In the model, the instantaneous intensity of incoming photosynthetically active radiation (PAR) is derived from the user-specified incoming short-wave radiation. We used the short-wave radiation time-series stemming from the NCEP Reanalysis. This is a global product at 2 degree resolution. Real-world incident short-wave radiation in the Marlborough Sounds may differ from this synthetic time-

series (because of local, perhaps seasonally varying, effects such as cloud-cover, atmospheric dust and sub-2-degree scale latitudinal variations).

PAR is almost entirely restricted to the visible spectrum, but this is composed of light of many wave-lengths. Even pure water absorbs some wave-lengths of visible light (e.g., red) much more strongly than others (e.g., green). The PAR attenuation coefficient represents an empirical measure of PAR absorption. Because it is PAR-based (rather than wave-length specific), its value tends to decline with increasing depth (as the residual PAR becomes increasingly concentrated in the weakly absorbed wave-lengths). The Fennel model does not break PAR down into multiple wave-length bands. Thus, it cannot take account of this subtlety. Our estimate of the PAR attenuation coefficient is based upon PAR measurements made from more than 8 m below the sea-surface. By this depth, all the strongly absorbed PAR wavelengths (which make up about 50% of the visible spectrum at the sea surface) have disappeared. The implication is that we are probably over-estimating the quantity of PAR which penetrates to 8 m and deeper. To some extent, this can be (and has been) accommodated through calibration of the initial slope of the photosynthesis-irradiance curve (to data from Queen Charlotte Sound during an earlier modelling exercise), but it is possible that this weakness in the model is responsible for some of its deficiencies with respect to reproducing the field data.

Finally, we have chosen to make long-term simulations on a grid having 200 m horizontal resolution. Long term simulations would have been prohibitively expensive on a finer grid (Table 2-1). 200 m resolution is approximately the size of the collective pen structures that comprise a fish farm. The biophysical model does not have sufficient resolution to properly represent the steep concentration gradients (for example, of ammonium and very large detritus) that will exist in the immediate environs of a farm. Specifically, it will exhibit excessive numerical dispersion such that it will tend to under-estimate concentrations very close to the farm, and, perhaps, over-estimate them slightly further afield. At greater distances (perhaps, >1 km), natural dispersion will have eroded the steep gradients so the excessive numerical dispersion is of lesser import and the simulated concentrations will be more reliable. If near-field concentrations are to be examined using this model, we would need to adopt a finer grid (e.g., 50 m or finer resolution) and restrict ourselves to simulating shorter calendar periods.

6.2 Model skill

We noted that the model appears to have lower skill for phytoplankton (carbon biomass and chlorophyll) than for other state-variables. We believe that this is a misleading conclusion that stems from an inconsistency between the phytoplankton communities represented in the model and in the data. The model community is 'total phytoplankton' (all size-classes). In contrast, the phytoplankton community sampled in the field is phytoplankton larger than approximately 2 μm (being the nominal pore size used when filtering for chlorophyll and the approximate minimum dimension of cells that can be reliably seen and measured under the optical microscope). The size-structure of the phytoplankton community within Pelorus Sound is not well known, but Safi and Gibbs (2003) report that between May 1999 and September 1999 (incl.), an average of 29% (range 8–65%) of the total phytoplankton chlorophyll was composed of phytoplankton <2 μm . Thus, it should come as no surprise that the total phytoplankton (as simulated in the model) exceeds the phytoplankton (>2 μm) measured in the field. Indeed, it would be disturbing if the modelled phytoplankton did not exceed the field phytoplankton. Thus, whilst our naïve comparison between modelled phytoplankton and field phytoplankton appear to indicate that the model reproduces phytoplankton less well than it reproduces other state-variables, we are satisfied that that is not the case.

Nonetheless, we certainly cannot claim that the model is reproducing the dynamics of any of the state-variables very well.

There are several reasons why this should not be overly surprising:

- we have made no attempt to calibrate the model to the field-data
- the model has no ability to mimic seasonal changes in phytoplankton community structure
- our Cook Strait boundary conditions are based upon scarce field data (just one year's worth of monthly measurements at only two depths). Furthermore, the raw-data were smoothed before being applied as boundary conditions
- the insolation intensities that are applied are not corrected for possible seasonal-scale variations in cloud-cover or seasonal and hour-by-hour variations in topographic shade (though the latter will be significant only in narrow parts of the Pelorus system)
- the hydrodynamic model is yielding summertime water temperatures which are a bit too low. Since phytoplankton and zooplankton physiology is temperature dependent, this (or possibly incorrect parameterisation of the temperature dependence) could have subtle influences upon emergent population growth rates and standing stocks. We emphasize that the temperature effects that are mediated through physiological changes are likely to be small. As a rule of thumb, the rates at which physiological processes proceed approximately double for each 10 °C temperature increment³⁸. Thus, the fact that simulated summertime water temperatures are approximately one °C too low implies that all temperature-dependent rates (primarily, ingestion (hence, egestion) and respiratory excretion) will be underestimated by about 5-10%
- the wind-fields that are applied derive from wind models that have low spatial resolution relative to widths of the Pelorus Channel. In combination with the steep topography, this implies that surface-flows and wind-driven mixing may not be well represented in the hydrodynamic model.

No specific numerical performance criteria have been set by which to assess the biophysical model's performance. In the context of this work, we are endeavouring to determine the relative changes induced by shellfish farming and fish-farming. With that in mind, it is appropriate to ask: 'does it matter that the model's performance is merely poor-moderate. For the reasons outlined in the following paragraph, we believe the answer is 'not greatly'.

Shellfish filter particulate matter out of the water column. The mussel model explicitly assumes that the quantity of water that each mussel pumps across its gills (the filtration apparatus) is independent of seston³⁹ concentration. Thus, the daily specific gross capture rate for seston is not influenced by the absolute quantity of seston in the water. On the other hand, the fraction of the captured seston that passes into the gut (rather than being rejected and converted to pseudo-faeces) is assumed to decline as the capture rate increases. That is, the relative quantity of captured seston that is rejected

³⁸ This is a rule-of-thumb that applies within an ectothermic organism's 'tolerable temperature range'. At more extreme (low or high) temperatures, the rates will drop rapidly toward zero due to temperature-induced damage to enzymes and cell structures. Differing functional forms (and/or coefficient values) have been adopted to describe the temperature dependencies of the various planktonic growth processes and mussel/fish ingestion, egestion and respiration processes (see appendices) but none imply a temperature-dependence that is markedly different from two-fold for ten degrees.

³⁹ Small, particulate organic matter in suspension within the water column (particulate organic detritus, phytoplankton and zooplankton)

as pseudofaeces (hence, returned to seston) increases with seston concentration. This means that even if the model over-predicts absolute seston concentration and, in consequence, initial capture of seston by mussels, much of 'excess' seston will be returned to the water-column (as pseudo-faeces-seston) rather than becoming incorporated into 'excess' mussel flesh (in effect, removed from the system).

Consequently, we infer that the model should be capable of adequately predicting near-field seston depletion levels if the near-field hydrodynamics are correct. Far-field change will be determined by a combination of many factors (hydrodynamics, plankton growth rates, detrital remineralization and settling rates etc.). The data do not allow us to determine whether the individual rates are close to being correct, but the fact that the far-field standing stocks are 'about right' on average offers some encouragement.

In the context of this model, fish farms are a source of nitrogen (as ammonium and particulate organic detritus). The rates of ammonium and detritus input are strongly correlated with the user-supplied fish-feed input rates, but also influenced by the parameterisation of the fish-physiology model. Those fish-feed input rates were based upon monthly rates provided to us by New Zealand King Salmon. We have recorded (but not reported) the derivative ammonium and detritus input rates calculated by our model and they are consistent with the prescribed inputs. It is worth emphasizing that the fish feed input rates reported by NZKS for the 2012/13 year were lower than they had been in earlier years – feed inputs had been reduced in order to reduce adverse impacts upon the seabed. Consequently, the nitrogen input rates for our *existing farms* scenario were lower than the maximum permitted by the consent conditions. By definition, that is appropriate for the *existing farms* simulation, but for the *approved farms* simulation it is less clear that this is appropriate. Nonetheless, for the most part, we chose to assume that the existing Pelorus fish farms would continue to operate in the same way as they did in 2012/13. The exceptions were Crail Li32 and Crail Li48. In reality, these were fallowed during 2012/13, but we assumed that they would be operating at their maximum permitted annual feed input rates in both the *existing* and *approved* farm scenarios.

The location (spatial and foodweb-level) and magnitudes of fish-farm induced change are dictated by myriad processes (currents, mixing, detrital sinking and mineralization rates, kinetics of plankton growth etc.). The hydrodynamic model has been shown to reproduce currents in the main stems of Pelorus moderately well (section 3.) The key biological processes governing how quickly (and how much) farm-derived nutrient is incorporated into the food chain are:

- Detrital denitrification rates.
- Detrital mineralization rates.
- Phytoplankton growth rates (particularly under nutrient-limiting conditions).

We have already established (section 5.2) that the denitrification rates are consistent with those measured in Pelorus Sound. Using the model output, it is possible to draw up nitrogen budgets. We do not present these in detail but the key conclusions are:

- At the whole of Pelorus scale, denitrification at the seabed (rather than export to Cook Strait) is the dominant means by which farm-derived nitrogen is removed from the system.

- Under the 'existing farms' scenarios, there is a net import of nitrogen from Cook Strait into Pelorus. Under the 'existing+approved farms' scenarios, there is a net export of nitrogen from Pelorus into Cook Strait.

We do not have data with which to validate any of the other biogeochemical rates predicted by our model but the coefficients that we have adopted to describe the various rate processes are typical of those seen in the water-column modelling literature. That said, though the specific detrital decay rates (0.01 d^{-1}) that we have adopted are typical of fresh plankton-derived material in the water-column (Enríquez, Duarte, Sand-Jensen 1993), they are high relative to those adopted when modelling the decay of fish-faeces in the seabed. For example, Brigolin, Pastres et al. (2009) adopted a value of 0.0027 d^{-1} . Since the bulk of farm-derived faeces will tend to arise in the summer, the implication is that too much of the farm-derived faecal nutrient will be mineralized during the summer (nutrient-limited months). Thus, the model may be over-estimating summertime fertilization potential. In reality, some of the faecal material which sinks to the seabed may not mineralize until autumn/winter. At that time of year, phytoplankton production will often be light-limited.

We have noted that, whilst the model appears to over-predict summertime, near-surface phytoplankton concentrations, there are reasons to believe that the over-prediction is not as bad as one might infer from a naïve comparison of simulation results (of total phytoplankton) and field-measurements (of phytoplankton $> 2 \mu\text{m}$). Caution must be applied whenever model results are compared with observations. It is not common practice to measure phytoplankton $< 2 \mu\text{m}$ in routine coastal plankton sampling (because the fine filters that are required to do so quickly become clogged with sediment and organic detritus).

6.3 Shifting baselines

In this work, we have regarded the 'existing conditions' simulation (i.e., EM-EF-WD) as our 'baseline'. Relative to that baseline, our modelling suggests that adding a (relatively) small number of additional mussel farms and three additional fish farms into the Pelorus system will induce water-quality changes that are small in comparison with present day seasonal variability. We have also shown that: (i) changes that might arise were benthic denitrification to be entirely lost⁴⁰ would be larger than those induced by the additional fish-farms and mussel farms that have been approved since 2010, (ii) that the changes arising from shifting from a notional 'no-farms' (i.e., pre-aquaculture development) stage to the present-day aquaculture stage ('existing farms') may have been several times greater than the changes that are predicted to be associated with the incremental addition of a relative small number of mussel farms [but a relatively large number of fish farms] between 2012 and a notional near-future state in which all of the approved mussel farms and forthcoming fish farms have been placed in the water.

In the model, all the simulated change that arises can legitimately be attributed to the aquaculture (we have changed nothing else in the model). In reality, however, the water-quality of the Sounds is likely to have been influenced by more than aquaculture expansion alone. Intensification of farming in some of the catchments may have caused nutrient inputs to climb. Probably more importantly, we believe that the nature of the seabed has changed dramatically over the past century. The evidence is laid out in detail within a recent report to Marlborough District Council (Handley 2015). In summary, there is substantial evidence that the surficial sediments of the seabed of Pelorus Sound

⁴⁰ An incomprehensibly unlikely event

used to be coarse grained sands, rock and biogenic reef (shellfish beds etc.). In contrast, the surficial seabed of Pelorus is now dominated by fine, soft, sediments. These are home to relatively few hard-bodied, long-lived organisms. We believe that the changes have been driven by a combination of increased sediment runoff from the catchment⁴¹, over-fishing of historical shellfish beds, and destruction of biogenic structures on the seabed by shellfish dredges and other near-bed trawl devices. Collectively, the changes to the seabed may have changed its capacity to mineralize organic nitrogen and denitrify ammonium and nitrate.

We have no robust, quantitative measurements of the water-quality of the Sounds prior to the development of any aquaculture (or loss of seabed habitat and change in catchment inputs). We might be tempted to use the present model as a means of hind-casting the state of the system. Arguably, doing so is legitimate, but it is certainly fraught with difficulties. Our naïve ‘no farms’ simulation is unlikely to yield an accurate indication of the historical state of Pelorus water quality. We made no attempt to change the catchment inputs of nitrogen (or water volume, temperature etc.,) to reflect a reversion to a native forest catchment. We did not adjust the light attenuation coefficient to reflect a (presumed) lower concentration of suspended sediments (but, perhaps, higher dissolved colours) in the pristine waters of the historical Sounds. We did not make any attempt to modify the fraction of depositing particulate organic nitrogen which denitrifies to N₂ and nor did we make any attempt to introduce a population of benthic shellfish into the notional ‘no farms’ simulation. In short, our ‘no farms’ simulation is too simple/naïve for its results to be plausibly regarded as being representative of the past state of Pelorus water-quality.

By drawing upon data from analogous pristine catchments, it might prove relatively easy to develop plausible⁴² (but always hypothetical) historical input loads for water and nutrient. We might also be able to develop plausible light attenuation coefficients and benthic grazing terms. Unfortunately, a fundamental scientific understanding of the manner in which seabed structure and faunal composition/activity etc., influence denitrification is poor. Developing plausible rules governing denitrification of sedimenting organic matter will certainly be much more difficult.

6.4 Implications of the biophysical modelling results: putting the changes in context

The MDC monitoring data indicate that near-surface nitrate concentrations vary more than ten-fold through the course of the seasons (Figure 5-1). Ammonium concentrations vary more than two-fold (Figure 5-2), phytoplankton and zooplankton concentrations vary five- to ten-fold (Figure 5-3 - Figure 5-10) and particulate detrital concentrations vary more than three-fold (Figure 5-4 - Figure 5-10). Even if one restricts attention to any one calendar month (taken from different years), the fluctuations can be substantial (compare pink and red circles and pink and blue triangles in Figure 5-4 - Figure 5-10). Unpublished historical data which NIWA gathered in Pelorus Sound indicates that a similar level of variability is also present there at a fortnightly time-scale.

Clearly, the predicted magnitudes of fish-farm-induced (or *denitrification associated*) change (relative to the *no farms* situation) are small relative to present-day natural variability. Furthermore, whilst we have chosen to focus upon seasonal-scale averages, inspection of time-series of instantaneous water-quality characteristics at a few specific locations have not revealed any relative changes which

⁴¹ as a consequence of initial clearance of the native forests and subsequent intermittent logging operations

⁴² In this context, not merely ‘defensible/likely’, but also ‘sufficiently precise/tightly constrained to be useful’

are dramatically larger than are evident in the seasonal averages. That is, the time-averaging is not obviously masking any extreme, but short-lived events that are driven by the farms.

The additional fish farms are predicted to increase summertime near-surface total phytoplankton standing stocks by 5–10% relative to the existing conditions. Even so, the model suggests that they will only rarely (and locally) exceed 5 mg Chl m^{-3} . Concentrations of that magnitude would probably not be high enough to begin to change the perceived colour of the water. Nor are they sufficiently high (for long enough and over sufficiently large areas) for the system to be classified as eutrophic.

In comparison with the magnitudes of natural variability, it is tempting to argue that 5-10% changes in (for example) phytoplankton standing stock are negligibly small — even when they persist for an entire season and over a large fraction of the Sound. That may be slightly naïve. Given sufficient time, a 5% change in resource availability could, in theory, permit a disproportionate change in consumer abundance.

Hansen, Bjørnsen and Hansen (1997) made an extensive review of the literature concerning the feeding and growth of zooplankton in the 2- 2000 μm size range (protozoa to large copepods). They concluded that the half saturation food concentration (food concentration at which an organism's ingestion rate is one half of maximal) was about 240 mg C m^{-3} (though, there is substantial between taxon variability – the individual estimates in the source literature vary more than ten-fold). Ingestion (and, by implication, individual growth rate) will rise approximately linearly with food concentration up to food concentrations around the half-saturation concentration. At higher concentrations, the scaling will be sub-linear. Seston concentrations measured at the MDC sampling stations range between 39 and 335 mg C m^{-3} (median 215 mg C m^{-3}). Thus, they are usually below the 'typical' half saturation coefficient. This implies that changes in seston abundance (whatever the cause) will tend to induce zooplankton individual-level growth rate changes that are approximately proportionate to those in the seston. In our model, mussels consume zooplankton as well as detritus and phytoplankton. Thus, the zooplankton must contend with direct grazing pressure as well as wide-spread mussel-induced depletion of their phytoplankton prey during the winter. Together with their much lower maximum specific growth rates, this explains why the zooplankton tend to suffer greater depletion than phytoplankton or detritus.

Higher in the foodweb, a correlation between annual-scale average seston abundance and mussel yields has been found in Pelorus Sound (Zeldis, Howard-Williams et al. 2008; Zeldis, Hadfield, Booker 2013). If we make a leap of faith and assume that the correlation is indicative of causation, this opens the possibility that fish-farming could be beneficial to mussel farmers – however, the benefit will be small. In the analyses by Zeldis, there was a roughly two-fold difference between the maximum and minimum annual average particulate N abundances. That was associated with a yield difference of approximately 30% (of the long term average). Thus, the change at the upper end of the foodweb was certainly not super-proportionate. Our model predicts that seston concentrations will increase by only a few percent during the summer months in response to additional fish farming. Furthermore, they revert to (fish)farm-free levels during the winter months. This regular 'reset' may introduce a 'bottleneck' that would limit the extent to which populations of short-lived organisms can develop a multi-annual response to regular summer-time enhancement.

There are no definitive/universal standards which state what an acceptable quantum of change might be for any water-column property in the context of aquaculture. In the Firth of Thames, a negotiation process led to agreement that, averaged over a year, mussel farming in the Wilson Bay Aquaculture Management Area A (Zeldis, Felsing, Wilson 2005):

- should not induce phytoplankton depletion that exceeded 25% over an area twice that of the AMA (the AMA has an area of approx. 1200 ha)
- should not induce phytoplankton depletion that exceeds 20% over more than 10% of the Firth's surface area.

The AMA concept has no direct equivalent in the Marlborough Sounds but, relative to the simulated present-day conditions, none of the 'future farms' scenarios yield time-averaged phytoplankton depletion in excess of 25%.

The New Zealand King Salmon Board of Inquiry stipulated several water quality standards that must not be broken (Final report Appendices 4-7). For example Appendix 4 section 51 stipulates:

51 The farm shall be operated at all times in such a way as to achieve the following qualitative Water Quality Standards in the water column:

- a To not cause an increase in the frequency or duration of phytoplankton blooms (i.e., chlorophyll a concentrations $\geq 5 \text{ mg/m}^3$)⁴³ [Note: water clarity as affected by chlorophyll a concentrations is addressed by this objective];*
- b To not cause a change in the typical seasonal patterns of phytoplankton community structure (i.e., diatoms vs. dinoflagellates), and with no increased frequency of harmful algal blooms (HAB"s) (i.e., exceeding toxicity thresholds for HAB species);*
- c To not cause reduction in dissolved oxygen concentrations to levels that are potentially harmful to marine biota [Note: Near bottom dissolved oxygen under the net pens is addressed separately through the EQS – Seabed Deposition];*
- d To not cause elevation of nutrient concentrations outside the confines of established natural variation for the location and time of year, beyond 250m from the edge of the net pens;*
- e To not cause a persistent shift from a mesotrophic to a eutrophic state;*
- f To not cause an obvious or noxious build-up of macroalgal (eg sea lettuce) biomass [Note to be monitored in accordance with Condition 80h].*

Three of these (a, d, e) can be addressed with our present model. First, we note that the Board appears to have adopted a threshold of $5 \text{ mg chl a m}^{-3}$ as indicative of eutrophy. The Consent Conditions do not make it clear, but referring back to the underlying evidence⁴⁴, it is clear that this should be interpreted as an annual average. The mere fact that one (or even several) samples yield chlorophyll concentrations in excess of $5 \text{ mg chl a m}^{-3}$ need not indicate that the system is in a eutrophied state.

Whilst the biophysical modelling indicates that a time-averaged threshold of $5 \text{ mg chl a m}^{-3}$ may be approached (even exceeded) at some locations during the summer period, it certainly doesn't indicate that it will be exceeded over a large fraction of Pelorus during the summer period. Furthermore, it will not be exceeded on a year-round basis (the relevant time-scale for this threshold). Our modelling spans a period of 500 days. It suggests that, over that time-span, fish farming (including the new farms) in Pelorus will not cause the system to shift into a eutrophied state. We cannot entirely refute the possibility of a longer-term evolution towards eutrophy (whether exhibited as persistently and substantially increased phytoplankton or substantial change elsewhere in the foodweb). Nonetheless, it is our current opinion that the combination of winter-

⁴³ The conditions do not stipulate which phytoplankton size fractions were to be included when calculating chlorophyll concentration

⁴⁴ The figure of $5 \text{ mg Chl a m}^{-3}$ appears to stem from evidence put forward by (Gillespie, P., Knight, B., MacKenzie, L. (2011) The New Zealand King Salmon Company Limited: assessment of environmental effects - water column: 79. citing Smith, V., Tilman, G., Nekola, J. (1999) Eutrophication: impacts of excess nutrient inputs on freshwater, marine, and terrestrial ecosystems. *Environmental Pollution*, 100(1-3): 179-196. and Wild-Allen, K., Herzfeld, M., Thomsen, P.A., Rosebrock, U., Parslow, J., Volkman, J.K. (2010) Applied coastal biogeochemical modelling to quantify the environmental impact of fish farm nutrients and inform managers. *Journal of Marine Systems*, 81: 134-147. 10.1016/j.marsys.2009.12.013).

time light limitation, relatively rapid flushing and benthic denitrification make it unlikely that the system will undergo extreme change in response to the levels of farming presently permitted in this system.

6.5 Biophysical modelling: summary of conclusions

- The model tends to predict overly high summertime surface-water phytoplankton concentrations in the baseline (existing conditions) scenario. Whilst we believe that the absolute concentrations increments that will arise when additional fish are added are small, the underlying 'baseline over-prediction' implies that : (a) the absolute summertime phytoplankton concentrations that may arise when more fish farms are added are probably also over-predicted by a similar increment, but (b) the relative concentration increments may be under-estimated.
- The 'no farms' simulation was run with the same (2012/2013-like) riverine, oceanic, climatic and seabed boundary conditions as the 'with farms' simulation. Thus, it may not be a good analogue for the true, historical no-farms situation. We know of no water-quality data stemming from prior to farm development. The earliest data that we know of stem from the mid/late 1990s (by which time there were already extensive mussel farms in Pelorus Sound). Analysis of those field data suggest that year-to-year variations in mussel yield were correlated with year-to-year variations in seston concentration and that those seston variations were best correlated with year-to-year climatic variations (El Nino/La Nina cycles). Whilst we have not made simulations for extreme El Nino or La Nina years, the discrepancy between model inferences and inferences drawn from the historical field data may indicate that the model is over-estimating the effects of mussels upon water-quality.
- Phytoplankton growth tends to be limited by low light intensities and short day-length during the winter months. During the summer months, it tends to be limited by a scarcity of nutrient (nitrogen). As a result of this difference, some of the effects of mussel and fish-farming differ between winter and summer months.
- Relative to the nominated baseline scenario (EM-EF-WD), a no mussel, existing fish with denitrification simulation (NM-EF-WD) yields:
 - Winter-time: lower concentrations of ammonium and nitrate but higher concentrations of particulate organic detritus (dead plankton etc.), phytoplankton and zooplankton. The largest changes in relative concentration are seen in Kenepuru Sound and the largest relative concentration changes are within the zooplankton. There, time-averaged near-surface winter-time seston⁴⁵ concentrations in the NM-EF-WD simulation are more than double those of the EM-EF-WD scenario (for zooplankton in Kenepuru, substantially more than double). The Beatrix/Crail/Clova system also exhibits similar (but smaller) changes.

⁴⁵ Collectively, phytoplankton, zooplankton and other small particulate material are referred to as seston. The mussels feed upon phytoplankton, zooplankton and detritus. They release detritus (as faeces and pseudo-faeces). Fish also generate faeces. None of this faecal and pseudo-faecal material is part of the seston because they sink very rapidly whereas, by definition, seston is supposed to be approximately neutrally buoyant.

- Summertime: lower concentrations of ammonium, nitrate, higher concentrations of detritus and zooplankton, but phytoplankton concentrations which are similar to (or lower than) those of the EM-EF-WD scenario. During summer, mussels convert particulate organic nitrogen (not directly exploitable by phytoplankton) to ammonium (directly exploitable by phytoplankton). Phytoplankton growth is normally nutrient limited during this time, but in the immediate vicinity of the mussel farms, phytoplankton (which survive passage through the farms) find a plentiful ammonium supply. This enables them to grow quickly – more than offsetting the losses that the population suffered to mussel grazing (the ‘excess accrued phytoplankton biomass being fuelled out of the detritus that was consumed). Once again, the largest changes are in Kenepuru Sound.
- Relative to the nominated baseline scenario (EM-EF-WD), a with mussel, no fish with denitrification simulation (EM-NF-WD) yields:
 - Winter-time: lower ammonium, nitrate and natural⁴⁶ detritus concentrations. With the exception of ammonium, the concentrations differ by less than approximately 1%. Phytoplankton and zooplankton concentrations that are almost identical to those of the EM-EF-WD scenario.
 - Summer-time: lower ammonium, nitrate, natural detritus, phytoplankton and zooplankton. The largest changes (declines in the absence of fish farms) are in Crail Bay (reflecting the presence of licensed farms in Crail Bay and Beatrix Bay and the slower flushing time of these bays in comparison with Waitata reach (which also harbours an existing fish farm at Waihinau Bay). Within Beatrix/Crail/Clova, time-averaged summertime phytoplankton concentration is predicted to be up to about 10% lower in the absence of fish farms. Zooplankton concentration is predicted to be up to about 15% lower.
- Turning to a comparison of the approved farms scenarios (AM-AF-WD) with the baseline (EM-EF-WD), the model predicts that the relatively few additional mussel farms present in the ‘approved farms’ scenarios (over and above those of the ‘existing farms’ scenario) induce water-quality changes that extend out to about bay-scale but amount to only a few percent of the simulated baseline (existing farms) concentrations. Changes are evident in nutrient (esp. ammonium) and seston concentrations. The changes include: increased ammonium concentrations in the vicinity of the farms and depressed concentrations of particulate organic detritus and zooplankton. During the winter, phytoplankton concentrations are slightly depressed by the additional mussel farms. During the summer, they are depressed in the immediate vicinity of the new mussel farms but can become slightly elevated further afield. The changes induced by these additional mussel and fish farms amount to a few percent of background concentrations. These are small relative to natural variability. For example, during winter, the incremental mussel grazing is predicted to induce local depletion of up to approximately 10% relative to the background/baseline (existing farms) simulation. In contrast, field data suggest that the extrema of phytoplankton population biomass can vary three or more fold over the course of a year. Indeed, it

⁴⁶ The small and large detritus classes of the model that receive dead plankton etc of the XL-detritus class that receives faeces and pseudo-faeces from the mussels and fish.

can sometimes fluctuate by almost that much over time-scales of weeks and space scales of km or less.

- The model predicts that fish farming induces effects will extend through the entire Pelorus system. The effects upon nutrients are more localized (and, there, more intense) than the effects upon phytoplankton, zooplankton or natural detritus. Relative to the 'existing conditions' (EM-EF-WD), the modelling suggests that the approved additional fish- and mussel farms will induce winter-time changes of <5% and summer-time changes of <15% at most. In winter, phytoplankton biomass will increase slightly in the main channel of central and inner Pelorus but decline within Crail/Clova/Beatrix Bays. In summer, they will increase throughout Pelorus. The greatest (albeit, still relatively small) changes will be in the vicinities of the new fish farms (i.e., in Beatrix/Crail/Clova Bays, and around Richmond/Waitata/Port Ligar).
- Wintertime light limitation acts as a 'bottleneck' which limits the response of short-lived organisms to the increased nutrient concentrations.
- The additional fish-farms boost the predicted (overly high relative to field data) by a small quantum. Whilst the predicted summertime, near surface, phytoplankton concentrations would be higher than is the norm for New Zealand coastal waters, they would not be higher than values that are intermittently recorded in our coastal waters. They would probably not be high enough to begin to change the perceived colour of the water. Thus, we do not believe that the concentrations of nutrients and phytoplankton associated with the fish-farming scenarios are alarmingly high (particularly as we know the model is over-predicting the 'existing condition' summertime concentrations). We reiterate that the EM-EF_WD model over-predicts summertime phytoplankton concentrations relative to field data.
- At the whole of Pelorus scale, the majority of the farm derived nitrogen is predicted to be lost through denitrification at the seabed of the Pelorus system rather than by export to Cook Strait.
- Whilst we believe that the inferences that we draw from our modelling are robust, we caution that almost no sensitivity trials have been undertaken to justify that belief. We therefore recommend that further sensitivity trials be undertaken to determine the degree to which the model predictions are robust against assumptions regarding:
 - Denitrification potential. The largest changes (relative to the existing situation) arose when we turned off benthic denitrification throughout the domain. We consider domain-wide loss of denitrification to be vanishingly unlikely. Nonetheless, denitrification can become suppressed when organic loading rates are very high. Thus, one might legitimately ask what happens if denitrification is suppressed only in those parts of the domain where organic loading is exceptionally high (i.e., under the fish farms)?
 - Light attenuation (what happens if we take better account of the differential attenuation of different wavelengths?; what happens if seasonal variations in cloud cover etc., is introduced?).

- Sensitivity to the phytoplankton half-saturation coefficient for uptake of ammonium and nitrate.
- Formulation of the zooplankton mortality term.
- The role of dissolved organic nitrogen (by how much would the system's dynamics change if we assumed that catchment- and ocean-derived DON was biologically active rather than inert?).
- Sensitivity to Cook Strait boundary conditions (can phase errors be reduced by removing the three month smoothing that was applied to the Port Gore data? What are the consequences of inter-annual variability in the extent to which Cook Strait water intrudes into Pelorus and/or in the water-quality characteristics of the intruding Cook Strait water?).
- Sensitivity to assumptions concerning the efficiencies with which mussels capture detritus and zooplankton out of the water (relative to the efficiency with which they capture phytoplankton).

The coupled hydrodynamic and NPZD/aquaculture model is complex and represents a real-world system which is difficult to study in detail. It is impractical to try to undertake detailed validation of all components of the model using field data. Instead, one must rely upon comparing relatively few emergent properties from the model against corresponding field data. On the other hand, one might endeavour to study some components of the model in more detail by comparing against detailed laboratory-scale data. For example, one might excise the Fennel NPZD model from the remaining (3D hydrodynamics and aquaculture) components and apply the resultant NPZD model to data from laboratory-scale incubation data. In such incubations, the physical conditions can be more tightly regulated and some rate processes can be directly measured. Excising the NPZD code would not be too onerous, but gathering the requisite incubation data would be a substantial undertaking. Short-term incubations of this general type have been undertaken (e.g., Carter 2004) but a detailed comparison of the NPZD model against such data is outside the scope of this work. Furthermore, a quick examination of the material presented within Carter (2004) suggests that only chlorophyll and nitrate concentrations were recorded. Thus, the study provides no information concerning many of the state-variables of the Fennel NPZD model.

7 Deposition modelling

7.1 Methods

We simulated the first-time deposition foot-prints of farm waste (faeces + uneaten food) using a particle-tracking model. Each virtual particle represents a 'parcel' of waste material (measured as grams of carbon). Particles were released on a continuous basis from random horizontal locations within the perimeter of each farm. Where the information was available, the perimeter that was adopted was that of the fish pens⁴⁷. In some cases (Port Ligar, Beatrix, Crail Li32 and Crail Li48) pen-perimeters were not available. For each of these farms, the perimeter was that of the licenced marine farm area. In these cases, the pens will occupy only a fraction of the area. At these farms, the model will: (a) underestimate maximal deposition under the pens and (b) over-estimate the area of the deposition footprint.

At release, each particle was also assigned a random initial depth between the sea-surface and 20 m below the surface. Subsequently, each particle moves under the influence of local-to-particle-currents, the intrinsic particle sinking velocity and turbulence.

The instantaneous local-to-particle currents were interpolated from an archive of 15 minute resolution hydrodynamic results generated by the 100 m resolution ROMS model. We adopted a sinking velocity of 5 cm s^{-1} (Brigolin, Pastres et al. 2009, and unpublished NIWA data). Turbulence was incorporated by adding a random velocity increment into each particle's equation of motion. The maximum absolute magnitude of this random term is proportional to the square-root of the estimated local dispersion coefficient. We assumed a horizontal dispersion coefficient of $1 \text{ m}^2 \text{ s}^{-1}$. Vertical dispersion was derived from the shear, with a Richardson Number correction term. This yielded dispersion coefficients in the range $10^{-5} - 10^{-1} \text{ m}^2 \text{ s}^{-1}$. We solved the resultant stochastic differential equation for particle motion by adopting Stratonovich Calculus and a second order Runge-Kutta method (Heun coefficients). For stochastic systems, this method is first-order strong convergent with respect to time-step. We adopted a time-step of 0.00025 d. This ensures that sinking (alone) cannot induce particles to pass through more than two layers within a single time-step. Thus, the particles will get to experience much (but not necessarily all) of any vertical variations in currents and mixing during their passage to the sea-bed.

At each farm, particles were released one at a time. The interval between particle releases was determined by the estimated daily rate of waste production ($\text{g C farm}^{-1} \text{ d}^{-1}$) and the nominal 'size' (g C) of each particle. The 'size' was chosen such that each farm generated 1000s of particles. The numbers of particles generated per farm ranged between approximately 15,000 (Waihinu) and 181,000 (Port Ligar). Each simulation spanned 30 simulated days. The farm-specific waste production rates were derived from monthly feed input rates ($\text{tonne feed farm}^{-1} \text{ month}^{-1}$), and an estimated carbon:feed weight fraction. This was derived from the C:dry weight ratios of protein, lipid and carbohydrate, and the typical proximate composition of salmon feed (Buschmann, Costa-Pierce et al. 2007). Refer to Table 7-1 for further details. For the Beatrix, Crail Li32, Crail Li48, Port Ligar, Richmond and Waitata farms, we applied monthly feed input rates that equated to 1/12 of the maximum permitted annual feed input rate. For Waihinu and Forsythe, we applied feed rates that were derived from projected feed schedules provided to us by NZKS. In both cases, the feed schedules included several months during which each farm would be empty. We derived an average

⁴⁷ Digitized from mooring plans provided to us by New Zealand King Salmon Ltd.

monthly input using only those months in which the farms would be occupied (ie, our deposition footprints for these farms are 'worst-case').

Table 7-1: Assumptions regarding composition of fish feed and assimilation of fish feed for deposition modelling.

Quantity	units	Value	Source
Fraction of ingested protein that is assimilated across gut wall	g assimilated g ⁻¹ ingested	0.90	(Buschmann, Costa-Pierce et al. 2007)
Fraction of ingested lipid that is assimilated across gut wall	g assimilated g ⁻¹ ingested	0.95	(Buschmann, Costa-Pierce et al. 2007)
Fraction of ingested carbohydrate that is assimilated across gut wall	g assimilated g ⁻¹ ingested	0.60	(Buschmann, Costa-Pierce et al. 2007)
Feed protein fraction	g protein g ⁻¹ feed	0.45	(Buschmann, Costa-Pierce et al. 2007)
Feed lipid fraction	g lipid g ⁻¹ feed	0.35	(Buschmann, Costa-Pierce et al. 2007)
Feed carbohydrate fraction	g carbohydrate g ⁻¹ feed	0.14	(Buschmann, Costa-Pierce et al. 2007)
Implied carbon content of feed	g C g ⁻¹ feed	0.47	
Implied assimilation efficiency for carbon	g assimilated g ⁻¹ ingested	0.82	
Assumed monthly feed input rate (Crail Bay Li48)	Tonne month ⁻¹	125	(1/12) of maximum permitted annual feed input rate
Assumed monthly feed input rate (Crail Bay Li32)	Tonne month ⁻¹	125	(1/12) of maximum permitted annual feed input rate
Assumed monthly feed input rate (Beatrix Bay)	Tonne month ⁻¹	208	(1/12) of maximum permitted annual feed input rate
Assumed monthly feed input rate (Richmond)	Tonne month ⁻¹	333	(1/12) of maximum permitted annual feed input rate
Assumed monthly feed input rate (Waitata)	Tonne month ⁻¹	500	(1/12) of maximum permitted annual feed input rate
Assumed monthly feed input rate (Waihinu)	Tonne month ⁻¹	172	Monthly average of projected input rates for period Dec '13 – Nov '14 [max permitted =333 month ⁻¹]

Quantity	units	Value	Source
Assumed monthly feed input rate (Port Ligar)	Tonne month ⁻¹	208	(1/12) of maximum permitted annual feed input rate
Assumed monthly feed input rate (Forsythe Bay)	Tonne month ⁻¹	215	Monthly average of projected input rates for period Nov '14-Sep '15. [max permitted =250 month ⁻¹]

7.2 Analysis and presentation of deposition model results

The location at which each particle first settled onto the seabed was recorded during the course of the simulation. Subsequently, all settlement locations were binned onto a 20 m resolution grid. This yields a bit-map of location-specific mass-accrual over the course of the 30-day simulation. Daily settlement rates are easily derived from that by dividing by the simulation length (30 d). We present the results as false-colour maps in which colour is indicative of the daily settlement rate (Figure 7-1).

Maximum deposition rates span the range 4.9 (Waitata) to 43.1 (Richmond) g C m⁻² (3.8–33.55 kg dry weight m⁻² year). The farm footprints range from 7.4 ha (Waihinau) to 32.5 ha (Port Ligar). Note, however that the latter is likely to be an over-estimate because particles were released from throughout the licenced area rather than from an (unknown) smaller pen-region within this area. The regions of maximum deposition are invariably within the farm perimeters and, in most cases, the deposition footprint is predicted to extend only approximately 100 m beyond the pen perimeter. Waitata and Richmond are exceptions. Their footprints are predicted to extend several hundred meters from the farm perimeters in the along-shore direction.

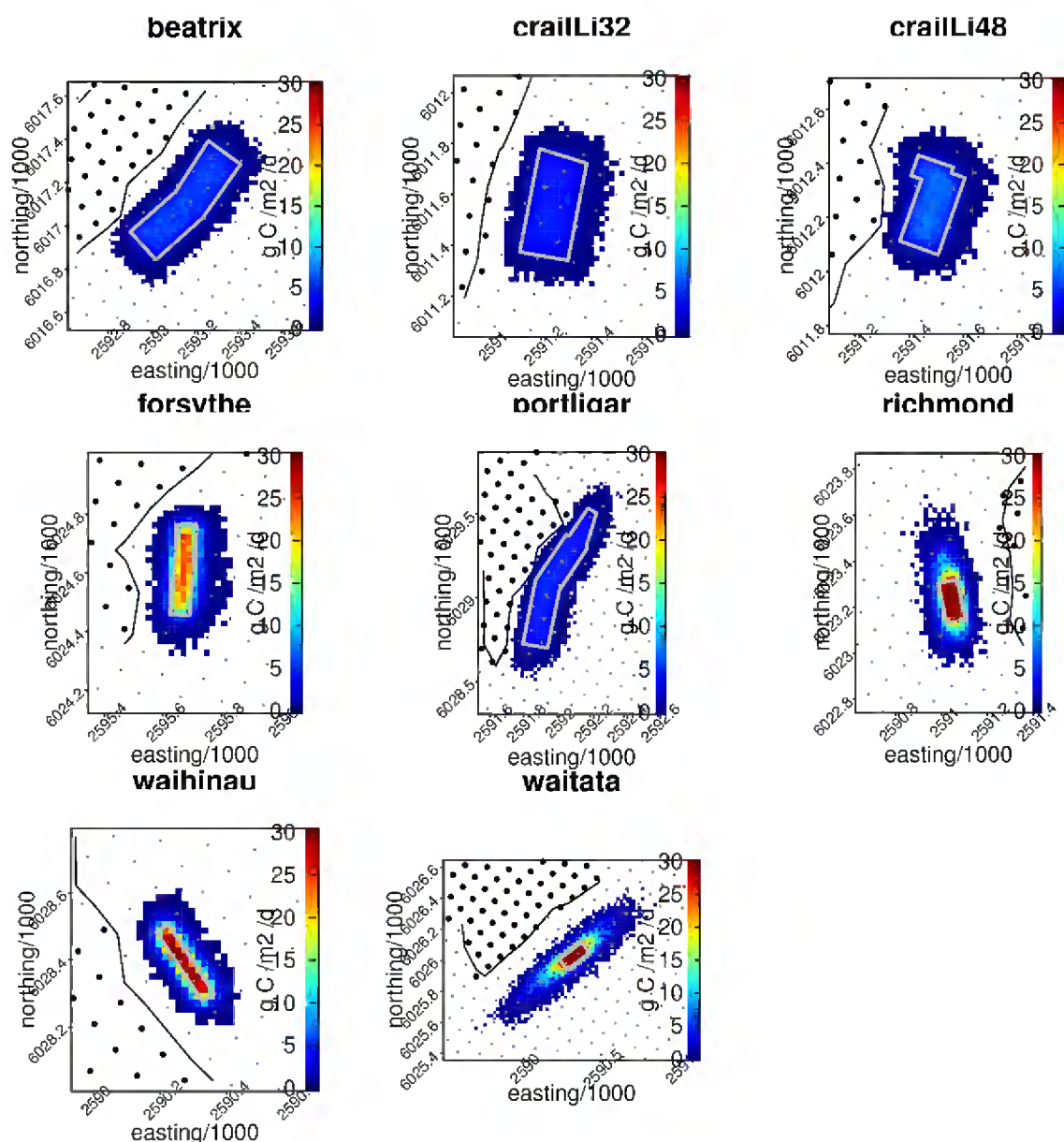


Figure 7-1: Maps of simulated daily deposition rates (g C / m^2) for each farm. Simulations span the period 13 July – 12 August 2012. Pixel colour is indicative of the deposition rate. The grey line indicates the perimeter of the particle-release regions in the model. The black line is the model's zero-depth contour. The black circles are the mid-points of the water-columns of the hydrodynamic model. Large circles denote 'water-columns' that are prescribed as being unwettable (i.e., permanently dry land). Small dots denote water-columns that are wettable.

7.3 Discussion

Deposition rates of $5\text{--}45 \text{ g C m}^{-2} \text{ d}^{-1}$ have been measured at the edge of the pen at Waihinai in Pelorus Sound (D. Morrissey, NIWA, unpublished data). Keeley, Cromey et al. (2013) used the DEPOMOD particle-tracking tool to simulate deposition under Waihinai farms. At the pen edges, his modelled deposition rates were about $10\text{--}20 \text{ kg solids m}^{-2} \text{ y}^{-1}$ ($13\text{--}26 \text{ g C m}^{-2} \text{ d}^{-1}$). Our model yields deposition rates at the Waihinai pen edges which are within this range (Figure 7-1). Similarly, at Forsythe, DEPOMOD predicts deposition rates of about $26 \text{ g C m}^{-2} \text{ d}^{-1}$ – as do our simulations. Keeley

and Taylor (2011) present DEPOMOD simulation results for Richmond and Waitata farms. Under the maximum consented loading scenarios, DEPOMOD predicts peak deposition rates of 19-22 kg solids $\text{m}^{-2} \text{y}^{-1}$ (24-28 g C $\text{m}^{-2} \text{d}^{-1}$) and a total footprint area of approximately 13 ha at Richmond. The corresponding figures from our simulations are 43 g C $\text{m}^{-2} \text{d}^{-1}$ and 17 ha. At Waitata, the corresponding DEPOMOD figures are 10-13 kg $\text{m}^{-2} \text{y}^{-1}$ (13-17 g C $\text{m}^{-2} \text{d}^{-1}$) and about 28 ha. Our model predicts 46 g C $\text{m}^{-2} \text{d}^{-1}$ and 28 ha. Overall, it is encouraging that the two models yield similar near pen deposition rates and deposition footprint areas, despite using very different sources of hydrodynamic forcing⁴⁸ and having been parameterized independently of one another.

There is no single 'critical' deposition rate which can be used as an unequivocal threshold value to distinguish between rates of deposition which will not induce changes to the seabed fauna and biogeochemical structure and those that will. Observations elsewhere suggest that the structure of the benthic faunal community can be expected to change when deposition rates exceed about 1–5 g C $\text{m}^{-2} \text{d}^{-1}$. For the purposes of monitoring NZKS farms, a so-called benthic Enrichment Score⁴⁹ system has been adopted. The details are discussed in a recent 'benthic quality standards and monitoring' document (Keeley, Gillard et al. 2014) but, broadly, scores ≤ 5.0 are deemed acceptable. Historical data from existing NZKS farms indicate that this threshold is increasingly likely to be exceeded when deposition rates come to exceed 5-10 kg solids $\text{m}^{-2} \text{y}^{-1}$ (6-12 g C $\text{m}^{-2} \text{d}^{-1}$) (Keeley, Cromey et al. 2013). On this basis, we suggest that few, if any of the farms will be able to operate at their maximum consented annual feed loads without breaching the agreed benthic standards. The aforementioned benthic quality standards document (Keeley, Gillard et al. 2014) provides an agreed framework by which stocking/feeding practices will be regulated in order to minimise the chances that farms will repeatedly violate the benthic standards.

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⁴⁸ data from a single current meter resolved into three depth bins and having no horizontal resolution vs horizontally resolved hydrodynamic model output resolved into 20 layers

⁴⁹ A weighted measure of the state of the seabed as indicated by the benthic faunal composition and abundance, and a variety of biogeochemical and characteristics of the bed.

9 Glossary of abbreviations and terms

ADCP	An acoustic Doppler current profiler, an instrument for measuring velocity profiles.
bathymetry	The process of measuring and analysing seafloor depth. A bathymetric data set is often informally called a bathymetry.
CTD	A conductivity-temperature-depth instrument, typically lowered and raised in the water to measure vertical profiles of temperature and salinity.
denitrification	A bacterially mediated process through which nitrate (NO_3^-) is converted to nitrous oxide gas (N_2O) and, in some circumstances, free nitrogen gas (N_2). Denitrification occurs under anoxic conditions. It tends to occur most rapidly in zones where oxic and anoxic areas are in close proximity to one another.
light-limited	The realizeable phytoplankton growth rate is limited by low intensities of ambient photosynthetically available radiation (PAR). The term is usually applied when considering growth averaged over a 24 hour period. Since PAR intensity declines with increasing distance below the sea-surface, near-bed waters are more likely to be light-limited than near-surface waters. Similarly, light-limitation is more likely during the winter than summer.
nitrification	A bacterially mediated process by which ammonium is converted to nitrate via nitrite. Nitrification requires the presence of free oxygen and is suppressed by PAR.
nutrient-limited	The realizeable phytoplankton growth rate is limited by low concentrations of nutrient in the water-column. The term is usually applied when considering growth averaged over a 24 hour period.
PAR	<u>Photosynthetically active radiation</u> : that part of the solar spectrum that plants (including phytoplankton) can harvest and utilize to drive photosynthesis.
stratified	When the water column is stratified, a surface layer of lower density water floats above a sub-surface layer of higher density water. The surface layer can be less dense because it is cooler or more salty than the sub-surface water, or a combination of both.

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Appendix A Mathematical description of the Fennel NPZD model

$$\begin{aligned} \frac{\partial Phy}{\partial t} = & \mu Phy - g_{Zoo} Zoo - m_{Phy} (Phy - PhyMIN)^+ - \tau(SDet + Phy) Phy - w_{Phy} \frac{\partial Phy}{\partial z} \\ & - \sum_{i=1}^{N_{muss}} Mus_i V_i Phy \psi_{Phy} \end{aligned}$$

The term $\sum_{i=1}^{N_{muss}} Mus_i V_i Phy \psi_{Phy}$ denotes the total local phytoplankton biomass loss rate ($\text{mmol N m}^{-3} \text{ d}^{-1}$) due to the mussels of each size-class i . Mus_i denotes the local concentration of mussels (mussels of size i class m^{-3}). V_i ($\text{m}^3 \text{ d}^{-1}$ mussel $^{-1}$) denotes the volume of water filtered across the gills and ψ_{Phy} ($0 < \psi_{Phy} \leq 1$) denotes the relative efficiency with which phytoplankton in the water passing over the gills is captured.

$$\begin{aligned} \mu = & 0.59 \mu_0 1.066^T \frac{\alpha I}{\sqrt{(0.59 \mu_0 1.066^T)^2 + (\alpha I)^2}} \left(\frac{NO3}{k_{NO3} + NO3} \right) \left(\frac{PhyIP}{k_{NH4} + NH4} \right) \left(\frac{NH4}{k_{NH4} + NH4} \right) \\ I = & I_0 \text{ par } e^{-z \left(K_w + \frac{K_{chl} \int_0^z chl(\zeta) d\zeta}{z} \right)} \end{aligned}$$

$$g_{Zoo} = g_{\max} \frac{Phy^2}{k_{Phy} + Phy^2}$$

$$\begin{aligned} \frac{\partial Chl}{\partial t} = & \rho_{Chl} \mu Chl - g_{Zoo} Zoo \frac{Chl}{Phy} - m_p (Chl - ChlMIN) - \tau(SDet + Phy) Chl - w_{Phy} \frac{\partial Chl}{\partial z} \\ & - \sum_{i=1}^{N_{muss}} Mus_i V_i Chl \psi_{Phy} \end{aligned}$$

$$\rho_{Chl} = \frac{\Theta_{\max} \mu Phy}{\alpha I Chl}$$

$$\begin{aligned} \frac{\partial Zoo}{\partial t} = & g_{Zoo} \beta Zoo - l_{bm} (Zoo - ZooMin)^+ - l_E \frac{Phy^2}{k_p + Phy^2} \beta Zoo - m_{Zoo} Zoo^2 \\ & - \sum_{i=1}^{N_{muss}} Mus_i V_i Zoo \psi_{Zoo} \end{aligned}$$

$$\begin{aligned} \frac{\partial SDet}{\partial t} = & g_{Zoo} (1 - \beta) Zoo + m_{Zoo} Zoo^2 + m_{Phy} Phy - \tau(SDet + Phy) SDet - r_{SDet} SDet \\ & - w_{SDet} \frac{\partial SDet}{\partial z} - \sum_{i=1}^{N_{muss}} Mus_i V_i SDet \psi_{SDet} \end{aligned}$$

$$\frac{\partial LDet}{\partial t} = \tau(SDet + Phy)^2 - r_{LDet} LDet - w_{LDet} \frac{\partial LDet}{\partial z} \sum_{i=1}^{N_{muss}} Mus_i V_i LDet \psi_{LDet}$$

$$\frac{\partial XLDet}{\partial t} = XLDet - w_{XLDet} \frac{\partial XLDet}{\partial z} - \sum_{i=1}^{N_{muss}} Mus_i V_i XLDet \psi_{XLDet} + \sum_{i=1}^{N_{fish}} faeces_i + uneatenfeed_i$$

$$\frac{\partial NO_3}{\partial t} = -\mu_{max} f(I) L_{NO_3} Phy + nNH_4$$

$$n = n_{max} \left(1 - \frac{I - I_0}{k_I + I - I_0} \right)^+$$

$$\frac{\partial NH_4}{\partial t} = -\mu_{max} f(I) L_{NH_4} Phy - nNH_4 + l_{BM} Zoo + l_E \frac{Phy^2}{k_P + Phy^2} \beta Zoo + r_{SDet} SDet + r_{LDet} LDet + \sum_{i=1}^{N_{muss}} excretion_i + \sum_{i=1}^{N_{fish}} excretion_i$$

Table 10-1: Coefficients of the Fennel module. Unless otherwise noted, the values are those specified in the code that forms a part of the ROMS distribution. The coefficients are listed by both their Fennel-paper and ROMS-code names. A few coefficients are present only in the ROMS-code. A little additional explanation for those is presented in the Comment column.

Coefficient (Fennel 2006)	Coefficient (ROMS code)	Description	Units	Value	Comment
K_w	AttSW	Light attenuation coefficient due to seawater and components other than chlorophyll	m^{-1}	0.21	MDC data for Secchi disk depth in Queen Charlotte converted to a diffuse light attenuation coefficient using a correlation between attenuation and Secchi disk depth established with data from Pelorus Sound (Vincent, Howard-Williams et al. 1989) and applying discount of approx. $0.02 m^{-1}$ to avoid 'double counting' of attenuation due to chlorophyll
K_{chl}	AttChl	Light attenuation coefficient for chlorophyll	$m^2 mg^{-1} chl$	0.02486	
par	PARfrac	Fraction of incident shortwave radiation that is photosynthetically active	-	0.43	
μ_0	Vp0	Temperature limited phytoplankton growth parameter	-	1.0	
I_0	I_thNH4	Radiation threshold for nitrification inhibition	$W m^{-2}$	0.0095	
k_I	D_p5NH4	Half saturation radiation for nitrification inhibition	$W m^{-2}$	0.1	

Coefficient (Fennel 2006)	Coefficient (ROMS code)	Description	Units	Value	Comment
n_{max}	NitriR	Maximum rate of nitrification	d^{-1}	0.05	
$1/k_{NO_3}$	K_NO3	Inverse half saturation for phytoplankton NO_3 uptake	$m^3 \text{ mmol}^{-1} \text{ N}$	2	
$1/k_{NH_4}$	K_NH4	Inverse half saturation for phytoplankton NH_4 uptake	$m^3 \text{ mmol}^{-1} \text{ N}$	2	
k_{phy}	K_Phy	Half saturation constant (squared) for zooplankton ingestion	$(\text{mmol N } m^{-3})^2$	2	
θ_{max}	Chl2C_m	Maximum Chl:phytoplankton carbon ratio	$\text{mg Chl } mg^{-1} \text{ C}$	0.0535	
NA	ChlMin	Minimum Chl:phytoplankton carbon ratio	$\text{mg Chl } mg^{-1} \text{ C}$	0.001	Additional coefficient present within ROMS. Chlorophyll background mortality falls to zero when the phytoplankton abundance falls below this value.
NA	PhyCN	Phytoplankton C:N ratio	$\text{mmol C } mmol^{-1} \text{ N}$	6.625	Additional coefficient present within ROMS. Required there for modelling of dissolved inorganic carbon and utilized in the mussel feeding model
$1/k_{NH_4}$	PhyIP	Phytoplankton, coeff governing NH_4 dependent inhibition of NO_3 uptake	$mmol^{-1} \text{ N}$	1.5	Note that the ROMS implementation of the Fennel model distinguishes two coefficients (K_NH4, PhyIP) that correspond to two different usages of the original Fennel model's coefficient k_{NH_4}
α	PhyIS	Initial slope of photosynthesis/irradiance curve	$(W \text{ } m^{-2} \text{ d})^{-1}$	0.0125	In the code, PhyIS is defined in the manner of (Fennel, Hetland et al. 2011) rather than that of (Fennel, Wilkin et al. 2006). The numeric value that we have adopted was derived by calibration. It is half of the ROMS-default, but the ROMS default value is towards the upper end of the (large) range cited in the literature (Fennel, Wilkin et al. 2006)
NA	PhyMin	Phytoplankton mortality guard threshold	$mmol \text{ N } m^{-3}$	0.001	Additional coefficient present within ROMS. Phytoplankton background mortality falls to zero when the phytoplankton abundance falls below this value.
m_{phy}	PhyMR	Phytoplankton specific 'background' mortality rate	d^{-1}	0.15	
β	ZooAE_N	Zooplankton assimilation efficiency for ingested nitrogen	-	0.75	

Coefficient (Fennel 2006)	Coefficient (ROMS code)	Description	Units	Value	Comment
l_{bm}	ZooBM	Zooplankton specific basal metabolic rate	d^{-1}	0.1	
NA	ZooCN	Zooplankton C:N ratio	$mmol\ C\ mmol^{-1}\ N$	6.625	Additional coefficient present within ROMS. Required there for modelling of dissolved inorganic carbon and utilized in the mussel feeding model
l_E	ZooER	Zooplankton specific excretion rate	d^{-1}	0.1	
g_{max}	ZooGR	Zooplankton maximum specific ingestion rate	d^{-1}	0.6	
NA	ZooMin	Zooplankton guard threshold for basal metabolism	$mmol\ N\ m^{-3}$	0.001	Additional coefficient present within ROMS. Zooplankton respiratory losses when zooplankton concentration falls below this threshold.
m_{Zoo}	ZooMR	Zooplankton specific mortality rate	d^{-1}	0.025	
R_{LDet}	LDERRN	Specific mineralization rate for N within large detritus	d^{-1}	0.01	Additional coefficient present within ROMS. Required there for modelling of dissolved inorganic carbon.
NA	LDERRC	Specific mineralization rate for C within large detritus	d^{-1}	0.01	
τ	CoagR	Specific rate for coagulation of small detritus and phytoplankton to large detritus	d^{-1}	0.005	
R_{SDet}	SDeRRN	Specific mineralization rate for N within small detritus	d^{-1}	0.01	
NA	SDeRRC	Specific mineralization rate for C within small detritus	d^{-1}	0.01	Additional coefficient present within ROMS. Required there for modelling of dissolved inorganic carbon
	XLDeRRN	Specific mineralization rate for N within very fast sinking detritus	d^{-1}	0.01	Additional coefficient to accommodate degradation of the new state-variable. Black (2012) [citing Attard (2010)] suggests $0.005 - 0.06\ d^{-1}$ for fish faeces and fish food. Giles and Pilditch (2006) estimated a rate of $0.16\ d^{-1}$ for the degradation of mussel-derived organic matter.
	XLDeRRC	Specific mineralization rate for C within very fast sinking detritus	d^{-1}	0.01	Additional coefficient to accommodate degradation of the new state-variable
w_{phy}	wPhy	Sinking velocity for phytoplankton	$m\ d^{-1}$	0.1	
w_{LDet}	wLDet	Sinking velocity for large detritus	$m\ d^{-1}$	1.0	

Coefficient (Fennel 2006)	Coefficient (ROMS code)	Description	Units	Value	Comment
w_{SDet}	wSDet	Sinking velocity for small detritus	$m\ d^{-1}$	0.1	
	wXLDet	Sinking speed for very fast detritus (faeces & pseudo-faeces)	$cm\ s^{-1} (m\ d^{-1})$	5 (4320)	This coefficient applies to the additional state-variable (very fast sinking detritus) that is not found in the original Fennel model. The sinking speed is based upon unpublished measurements of faecal sinking speeds for material from Chinook salmon farmed in the Marl Sounds. It towards the upper range of published values for fish and mussel faecal sinking speeds (Cromey, Nickell et al. 2009; Giles, Broekhuizen et al. 2009; Reid, Liutkus et al. 2009)

Table 10-2: Coefficients required to link the Fennel NPZD model and the Ren mussel physiology model.

The coefficients in this Table are not found in either of the original Fennel or Ren models but they are required in order to allow the models to be coupled. The coefficients used in our implementation of the mussel physiology are those specified within Ren & Ross (2005) or Ren et al. (2010).

Coefficient	Description	Units	Value	Comment
LDeCN	C:N ratio for large detritus	mmol C mmol ⁻¹ N	6.625	Assumed, but consistent with Fennel model C:N ratios of zooplankton and phytoplankton and the assumption that detrital C & N mineralize at the same rates
SDeCN	C:N ratio for small detritus	mmol C mmol ⁻¹ N	6.625	Assumed, but consistent with Fennel model C:N ratios of zooplankton and phytoplankton and the assumption that detrital C & N mineralize at the same rates
SIS	Concentration of suspended inorganic sediment	mg ash weight m ⁻³	2000	Marlborough District Council water quality samples from Queen Charlotte Sound
ψ_{Phy}	Relative search volume of mussels for phytoplankton	-	1.0	By definition
ψ_{Zoo}	Relative search volume of mussels for zooplankton	-	1.0	(Zeldis, Robinson et al. 2004)
ψ_{LDet}	Relative search volume of mussels for large detritus	-	1.0	Assumed, but consistent with (Zeldis, Robinson et al. 2004)
ψ_{SDet}	Relative search volume of mussels for small detritus	-	1.0	Assumed, but consistent with (Zeldis, Robinson et al. 2004)
ψ_{XLDet}	Relative search volume of mussels for small detritus	-	0.0	Assumed
PhyDWN	Dry weight to nitrogen ratio for phytoplankton	g DW mmol ⁻¹ N	1.02	(Bowie, Mills et al. 1985)
ZooDWN	Dry weight to nitrogen ratio for phytoplankton	g DW mmol ⁻¹ N	0.89	(Beers 1966)
LDeDWN	Dry weight to nitrogen ratio for large detritus	g DW mmol ⁻¹ N	1.0	Assumed, chosen to lie between the corresponding ratios for phytoplankton and zooplankton (closer to the former)
SDeDWN	Dry weight to nitrogen ratio for small detritus	g DW mmol ⁻¹ N	1.0	Assumed, chosen to lie between the corresponding ratios for phytoplankton and zooplankton (closer to the former)

Appendix B Mathematical description of the mussel farm model

The full Ren et al. (2010) mussel growth model includes explicit dynamic descriptions of the rates of change of mussel energy reserves and structural volume. In his model, the reserve:structure ratio can vary through time (it provides an index of mussel condition or level of starvation). Some of the physiological rates are influenced by the ratio. We do not go to these lengths. The mussels of our population are described only in terms of numbers per length class. Length and structural volume are closely related, but instantaneous length provides no information about mussel condition. For the purposes of calculating all physiological rates, we assume that our mussels have replete reserves.

An individual mussel is defined by its shell length (M_L , mm). In turn, this defines various body-weight characteristics. The whole animal wet-weight (inclusive of shell, gram) is denoted M_{WW+S} :

$$M_{WW+Shell} = 0.00025 M_L^{2.726}$$

The wet-weight exclusive of shell (gram) is:

$$M_{WW} = 0.32 M_{WW+Shell}$$

The dry weight (exclusive of shell, gram) is:

$$M_{DW} = 0.2 M_{WW}$$

We assume that our mussels have replete reserves, and that reserves amount to 40% of the dry body mass (exclusive of the shell). Thus, the dry weight mass (gram) of structural tissue (ie proteins, carbohydrates etc., which once laid down, cannot be remobilized to meet energetic demands etc.,) is:

$$M_S = 0.6 M_{DW}$$

and, the dry weight mass (gram) of mussel reserve materials is:

$$M_R = M_{DW} - M_S = 0.4 M_{DW}$$

The energy content (J) of these reserves is

$$E = \frac{1000 M_R}{\mu_E}$$

The biovolume (M_V) of the structural material is

$$M_V = \frac{M_S}{\rho}$$

In the original Ren model, the maximum energy reserves (J cm⁻³) are denoted $[E_m]$. In our derivation of this model, we assume $E = [E_m] M_V$.

$$M_V = \frac{M_{DW}}{0.2} = M_{WW}$$

The energy content of the mussel (Joules, exclusive of shell) is

$$M_J = 1000(M_S \mu_S + M_R \mu_R)$$

As noted previously, we assume the mussels have replete reserves, so

$$M_R \mu_R = \frac{M_J - 1000 M_S \mu_S}{1000} = 2600 M_V$$

Mussels are assumed to consume seston. We measure its abundance (as perceived by the mussels) as: carbon (S_C , mg C m⁻³), nitrogen (S_N , mg N m⁻³) and dry weight (S_{DW} , mg dry weight m⁻³). Seston is assumed to comprise of small and large detritus, phytoplankton and zooplankton and suspended inorganic matter. The carbon (S_C , mmol C m⁻³), nitrogen (S_N , mmol N m⁻³), dry-weight (S_{DW} , mg m⁻³) and energy concentrations (S_J , J m⁻³) of perceived seston are given by:

$$S_C = \text{Phy} \cdot \varphi_{C:N}^{\text{Phy}} \psi_{\text{Phy}} + \text{Zoo} \cdot \varphi_{C:N}^{\text{Zoo}} \psi_{\text{Zoo}} + \text{LDet} \cdot \varphi_{C:N}^{\text{LDet}} \psi_{\text{LDet}} + \text{SDet} \cdot \varphi_{C:N}^{\text{SDet}} \psi_{\text{SDet}}$$

$$S_N = \text{Phy} \psi_{\text{Phy}} + \text{Zoo} \psi_{\text{Zoo}} + \text{LDet} \psi_{\text{LDet}} + \text{SDet} \psi_{\text{SDet}}$$

$$S_{DW} = \text{Phy} \cdot \varphi_{DW:N}^{\text{Phy}} \psi_{\text{Phy}} + \text{Zoo} \cdot \varphi_{DW:N}^{\text{Zoo}} \psi_{\text{Zoo}} + \text{LDet} \cdot \varphi_{DW:N}^{\text{LDet}} \psi_{\text{Zoo}} + \text{SDet} \cdot \varphi_{DW:N}^{\text{SDet}} \psi_{\text{Zoo}} + \text{SIS}$$

$$S_J = \text{Phy} \cdot \varphi_{C:N}^{\text{Phy}} \cdot \varphi_{J:C}^{\text{Phy}} \psi_{\text{Phy}} + \text{Zoo} \cdot \varphi_{C:N}^{\text{Zoo}} \cdot \varphi_{J:C}^{\text{Zoo}} \psi_{\text{Zoo}} + \text{LDet} \cdot \varphi_{C:N}^{\text{LDet}} \cdot \varphi_{J:C}^{\text{LDet}} \psi_{\text{LDet}} + \text{SDet} \cdot \varphi_{C:N}^{\text{SDet}} \cdot \varphi_{J:C}^{\text{SDet}} \psi_{\text{SDet}}$$

The volume of water pumped across the mussel gill surface is:

$$V = U_{mm} M_V^{2/3} f(K)$$

Where $f(K)$ denotes the temperature dependence function (temperature in Kelvin)

$$f(K) = k_{T0} e^{\left(\frac{T_A}{T_0} - \frac{T_A}{K}\right)} \left[1 + e^{\left(\frac{T_{AL}}{K} - \frac{T_{AL}}{T_L}\right)} + e^{\left(\frac{T_{AH}}{T_H} - \frac{T_{AH}}{K}\right)} \right]^{-1}$$

The quantities of phytoplankton, zooplankton, large detritus, small detritus and energy captured on the gills are:

$$C_{\text{Phy}} = V \text{Phy} \psi_{\text{Phy}}$$

$$C_{\text{Zoo}} = V \text{Zoo} \psi_{\text{Zoo}}$$

$$C_{\text{LDet}} = V \text{LDet} \psi_{\text{LDet}}$$

$$C_{\text{SDet}} = V \text{SDet} \psi_{\text{SDet}}$$

$$C_J = V \left(\text{Phy} \psi_{\text{Phy}} \varphi_{C:N}^{\text{Phy}} \varphi_{J:C}^{\text{Phy}} + \text{Zoo} \psi_{\text{Zoo}} \varphi_{C:N}^{\text{Zoo}} \varphi_{J:C}^{\text{Zoo}} + \text{LDet} \psi_{\text{LDet}} \varphi_{C:N}^{\text{LDet}} \varphi_{J:C}^{\text{LDet}} + \text{SDet} \psi_{\text{SDet}} \varphi_{C:N}^{\text{SDet}} \varphi_{J:C}^{\text{SDet}} \right)$$

Of this material, a fraction is lost as pseudo-faeces. The remainder passes into the gut. The fraction passing into the gut is given by:

The rate at which energy is assimilated across the gut wall is

$$A_J = V \frac{S_C}{S_C + H_{pm}} p_{A_{\max}}$$

The rates of carbon and nitrogen assimilation are:

$$A_C = \frac{V(Phy\psi_{Phy}\varphi_{C:N}^{Phy} + Zoo\psi_{Zoo}\varphi_{C:N}^{Zoo} + LDet\psi_{LDet}\varphi_{C:N}^{LDet} + SDet\psi_{SDet}\varphi_{C:N}^{SDet})}{C_J} I_J$$

$$A_N = \frac{V(Phy\psi_{Phy} + Zoo\psi_{Zoo} + LDet\psi_{LDet} + SDet\psi_{SDet})}{C_J} I_J$$

Material which is not assimilated across the gut wall is lost as faeces and pseudo-faeces and passes into the large-detrital pool.

The mussel energy expenditure rate ($J \text{ mussel}^{-1} \text{ d}^{-1}$) is made up of a basal term (p_M) and a growth-and-filtration-related term (p_g).

$$p_M = \left(\frac{[E]}{[E_G] + \kappa[E]} \right) f(K)[p_m]M_V$$

$$p_g = \left(\frac{[E]}{[E_G] + \kappa[E]} \right) [E_G]p_{A_{\max}}M_V^{2/3}$$

The mussel carbon respiration rate (E_C , $\text{mmol CO}_2\text{-C mussel}^{-1} \text{ d}^{-1}$) is:

$$E_C = \max \left[A_C - \frac{p_M + p_g}{\mu_R}, A_C - A_N\varphi_{C:N}^{Mus} \right]$$

The mussel nitrogen excretion rate (E_N , $\text{mmol NH}_4\text{-N mussel}^{-1} \text{ d}^{-1}$) rate is:

$$E_N = A_N - \frac{1}{\varphi_{C:N}^{Mus}} \left[A_C - \frac{p_M + p_g}{\mu_R} \right]$$

Symbol	Description	Units	Value	Comment
	Scaling coefficient relating whole animal wet weight (incl. of shell) to shell length	$\text{g mm}^{-2.726}$	0.00025	(Hickman 1979)
	Exponent in wet-weight:length relationship	-	2.76	(Hickman 1979)
	Fraction of whole animal wet weight that is not shell	-	0.32	
	dry weight: wet weight ratio of mussel soft tissue	-	0.2	
	Structural tissue dry weight mass/soft tissue dry weight mass for a well fed mussel	-	0.6	
μ_S	Energy density of mussel structural tissue	J mg^{-1} structural dry weight		
μ_R	Energy density of mussel reserve tissue	J mg^{-1} reserve dry weight		

Symbol	Description	Units	Value	Comment
$\phi_{C:N}^{Phy}$	C:N ratio of phytoplankton	mol C / mol N		
$\phi_{C:N}^{Zoo}$	C:N ratio of zooplankton	mol C / mol N		
$\phi_{C:N}^{LDet}$	C:N ratio of large detritus	mol C / mol N		
$\phi_{C:N}^{SDet}$	C:N ratio of small detritus	mol C / mol N		
$\phi_{DW:N}^{Phy}$	C:N ratio of phytoplankton	g dry weight / mol N		
$\phi_{DW:N}^{Zoo}$	C:N ratio of zooplankton	g dry weight / mol N		
$\phi_{DW:N}^{LDet}$	C:N ratio of large detritus	g dry weight / mol N		
$\phi_{DW:N}^{SDet}$	C:N ratio of small detritus	g dry weight / mol N		
$\phi_{J:C}^{Phy}$	Energy density of phytoplankton	J / mmol C		
$\phi_{J:C}^{Zoo}$	Energy density of zooplankton	J / mmol C		
$\phi_{J:C}^{LDet}$	Energy density of large detritus	J / mmol C		
$\phi_{J:C}^{SDet}$	Energy density of small detritus	J / mmol C		
ψ^{Phy}	Mussel filtration efficiency for phytoplankton	m ³ m ⁻³		
ψ^{Zoo}	Mussel filtration efficiency for zooplankton	m ³ m ⁻³		
ψ^{LDet}	Mussel filtration efficiency for large detritus	m ³ m ⁻³		
ψ^{SDet}	Mussel filtration efficiency for small detritus	m ³ m ⁻³		
H_{pm}	Half saturation seston concentration	mmol C m ⁻³	295/12	
$p_{A_{max}}$	Maximum surface area specific assimilation rate	J cm ⁻² d ⁻¹		
ρ	Biovolume-specific concentration of structural materials	g structural cm ⁻³ biovolume	0.2	

Appendix C Mathematical description of the fish farm model

Stigebrandt derived a model for salmon growth that is based upon energy conservation. Fish size is expressed as live weight (W , gram), and energy content (Q , Joules). The energy density of fish flesh (C_{fi} , J g⁻¹ live weight) is assumed to be constant.

$$Q = WC_{fi}$$

The maximal fish growth rate (G_{\max} , g live weight fish⁻¹ d⁻¹) is assumed to scale allometrically with fish weight and exponentially with temperature (T , Celsius).

$$G_{\max} = aW^b e^{\tau T}$$

The realized ingestion rate (Q_r , J fish⁻¹ d⁻¹) is the lesser of the per-capita feed provision rate (Q_{feed} , J fish⁻¹ d⁻¹) or the maximal ingestion rate ($Q_{r\max}$, J fish⁻¹ d⁻¹, to be defined in greater detail later)

$$Q_r = \min(Q_{feed}, Q_{r\max}) aW^\gamma e^{\tau T}$$

The feed is deemed to consist of a water fraction (F_w , g water g⁻¹ feed), a protein fraction (F_p , g protein g⁻¹ feed), a lipid fraction (F_l , g lipid g⁻¹ feed) and a carbohydrate fraction (F_c , g carbohydrate g⁻¹ feed). The energy densities (J g⁻¹ substrate) for lipid and carbohydrate are denoted C_l and C_c respectively. For protein, we define two energy densities. $C_p^{NO_3}$ denotes the energy density if the protein is fully catabolised to yield NO₃ as the nitrogenous end-product. $C_p^{NH_4}$ denotes the (smaller) energy density that arises when protein is catabolised to yield ammonium as an end-product. The energy density of food is defined to be:

$$\delta = F_p C_p^{NO_3} + F_l C_l + F_c C_c$$

The fractional contributions of protein, lipid and carbohydrate to the total ingested energy are:

$$E_p^{NO_3} = \frac{F_p C_p^{NO_3}}{\delta}, E_p^{NH_4} = \frac{F_p C_p^{NH_4}}{\delta}, E_l = \frac{F_l C_l}{\delta} \text{ and } E_c = \frac{F_c C_c}{\delta}$$

A fraction of the ingested energy is lost in faeces. The loss rate (Q_f , J fish⁻¹ d⁻¹) is determined by the assimilation efficiencies for protein (A_p , dimensionless), carbohydrate (A_c) and lipid (A_l) and by the fractional contributions which each makes to total energy ingestion.

$$Q_f = Q_r \left((1 - A_p) E_p^{NO_3} + (1 - A_l) E_l + (1 - A_c) E_c \right)$$

The process of breaking proteins, lipids and carbohydrates into simpler molecules and assimilating those across the gut wall incurs an energy expenditure (so-called specific dynamic action, (Q_{SDA} , J fish⁻¹ d⁻¹). The SDA for protein amounts to 30% of the assimilated protein energy whilst the SDAs for lipid and carbohydrate amount to 5% of their respective energy assimilation rates:

$$Q_{SDA} = Q_r \left(0.3 A_p E_p^{NO_3} + 0.05 (A_c E_c + A_l E_l) \right)$$

A fish is assumed to use dietary lipid and carbohydrate preferentially to fuel its energy demands (thereby conserving as much nitrogen as possible to synthesize new proteins). Nonetheless, when a fish assimilates more nitrogen than it requires to meet the nitrogen demands associated with building new flesh, it uses the excess protein to meet energetic expenditure. Similarly, when the total energy assimilation rate is insufficient to meet the basal energy demand, the fish is assumed to

meet the deficit by catabolising lipid, carbohydrate and protein at rates which maintain a fixed proximate body composition. $E_p^{NO_3}$ is based upon full oxidation to NO_3 , but fish catabolise proteins only to NH_4^+ . Thus, account must be taken of the energy that is lost as NH_4^+ when protein is catabolised. The result is an additional growth-related energy loss (Q_N , J fish⁻¹ d⁻¹)

$$Q_N = \frac{E_p^{NO_3} - E_p^{NH_4}}{E_p^{NO_3}} C_p \left(F_p A_p \frac{Q_r}{\delta} - P_p \frac{dW}{dt} \right)$$

The energetic cost of growth (net accrual of new fish flesh; Q_g , J fish⁻¹ d⁻¹) is assumed to be proportional to the rate of growth.

$$Q_g = C_{fi} \left[\frac{dW}{dt} \right]^+$$

Basal energetic costs (Q_s , J fish⁻¹ d⁻¹) are assumed to scale allometrically with fish weight and exponentially with temperature.

$$Q_s = \alpha W^\gamma e^{\tau T}$$

In the original Stigebrandt model, locomotory costs (Q_l , J fish⁻¹ d⁻¹) were set to zero because basal metabolism was explicitly assumed to include a locomotory component. In our implementation, we have retained an explicit locomotory term (set proportional to the basal respiration – which explicitly excludes locomotion)

$$Q_l = \vartheta Q_s$$

Since farmed salmon are usually harvested before reaching sexual maturity, we assume that the energetic costs of gamete synthesis (Q_p , J fish⁻¹ d⁻¹) are zero.

Collectively, the expressions for basal metabolism, maximal growth rate, maximal ingestion rate, digestive efficiencies, specific dynamic action, and protein catabolism efficiency imply an expression for the maximum ingestion rate (J fish⁻¹ d⁻¹)

$$Q_{r_{\max}} = \frac{\left[\alpha W^\gamma + a W^b \left(C_{fi} - \frac{E_p^{NO_3} - E_p^{NH_4}}{E_p^{NO_3}} C_p P_p \right) \right] e^{\tau T}}{1 - \left((1 - A_p) E_p + (1 - A_l) E_l + (1 - A_c) E_c \right) - \left(0.3 A_p E_p^{NO_3} + 0.05 (A_c E_c + A_l E_l) \right) - \frac{E_p^{NO_3} - E_p^{NH_4}}{E_p^{NO_3}} A_p E_p}$$

The rate of change of energy content $\frac{dQ}{dt}$ (J fish⁻¹ d⁻¹) is given by the difference between the rates of energy ingestion (Q_r) and energy loss through: faeces Q_f , catabolism of protein ingested in excess of growth requirements (Q_N), basal metabolism (Q_s), locomotory metabolism (Q_l), specific dynamic action, energy expended in synthesis of new flesh (Q_g) and energy expended in synthesis of gametes (Q_p).

$$\frac{dQ}{dt} = Q_r - Q_f - Q_N - Q_s - Q_l - Q_{SDA} - Q_g - Q_p = C_{fi} \frac{dW}{dt}$$

Since Q_N and Q_g are dependent upon $\frac{dW}{dt}$, the equation does not have an analytic solution. We use the bisection method to calculate a realized instantaneous value for $\frac{dQ}{dt}$ that satisfies this equation. Conceptually similar equations can be set up for carbon. Again, we use the bisection method to solve that equation. The final realized growth rate (which may be negative) is the lesser of the two growth rates (expressed in energy units). Realized carbon, nitrogen etc., uptake and release fluxes are then calculated on the basis of that growth rate. Oxygen demand can be derived from the calculated assimilation rates of protein, carbohydrate and lipid, the realized fish growth rate and the respiratory quotient of each substrate. (Buschmann, Costa-Pierce et al. 2007)

Table 10-3: Coefficients for the fish physiology module. WW: wet weight. The majority of coefficients are derived from data concerning Atlantic salmon. Chinook salmon is the species that farmed in the Marlborough Sounds).

Coefficient (Stigebrandt 1999)	Coefficient (ROMS code)	Description	Units (Stigebrandt) [ROMS]	Value	Comment
$C_p^{NO_3}$		Energy protein when fully oxidized to nitrate	J g ⁻¹	23.0x10 ³	http://www.fao.org/docrep/003/aa040e/aa040e08.htm
$C_p^{NH_4}$		Energy density of protein when catabolized to ammonium	J g ⁻¹	19.0x10 ³	(Schmidt-Nielsen 1982)
C_l		Energy density of lipid	J g ⁻¹	39.33x10 ³	(Schmidt-Nielsen 1982)
C_c		Energy density of carbohydrate	J g ⁻¹	17.57x10 ³	(Schmidt-Nielsen 1982)
	FeedWaterFrac	Energy density of ingested carbohydrate	g water g ⁻¹ feed	0.085	(Buschmann, Costa-Pierce et al. 2007)
F_p	FeedProteinFrac	Protein content of the fish feed	g protein g ⁻¹ feed	0.45	(Buschmann, Costa-Pierce et al. 2007); http://en.wikipedia.org/wiki/Chinook_salmon
F_l	FeedLipidFrac	Lipid content of the fish feed	g lipid g ⁻¹ feed	0.22	http://en.wikipedia.org/wiki/Chinook_salmon ; but see (Buschmann, Costa-Pierce et al. 2007) – which suggests 0.35
F_c	FeedCarbFrac	Carbohydrate content of fish feed	g carbohydrate g ⁻¹ feed	0.14	http://en.wikipedia.org/wiki/Chinook_salmon ; but see (Buschmann, Costa-Pierce et al. 2007) – which suggests 0.10
	FishWaterFrac	Water content of the fish feed	g water g ⁻¹ fish	0.75	(Shearer, Åsgård et al. 1994)
P_p	FishProteinFrac	Protein content of the fish	g protein g ⁻¹ fish	0.14	(Shearer, Åsgård et al. 1994)
P_l	FishLipidFrac	Lipid content of the fish	g lipid g ⁻¹ fish	0.10	(Shearer, Åsgård et al. 1994)
P_c	FishCarbFrac	Carbohydrate content of fish	g carbohydrate g ⁻¹ fish	0.015	(Shearer, Åsgård et al. 1994)

Coefficient (Stigebrandt 1999)	Coefficient (ROMS code)	Description	Units (Stigebrandt) [ROMS]	Value	Comment
δ	Derived property (see right)	Energy density of fish feed	J g ⁻¹	δ $= F_p C_p^{NO_3}$ $+ F_l C_l$ $+ F_c C_c$	$1 \leq F_p + F_l + F_c$, allowing that fish feed may have a small water content
C_{fi}	Derived property (see right)	Energy density of live fish	J g ⁻¹	C_{fi} $= P_p C_p^{NO_3}$ $+ P_l C_l$ $+ P_c C_c$	
A_p	FishAssimEfficProt	Assimilation efficiency for protein content of fish feed	-	0.9	(Buschmann, Costa-Pierce et al. 2007)
A_l	FishAssimEfficLipid	Assimilation efficiency for lipid content of fish feed	-	0.95	(Buschmann, Costa-Pierce et al. 2007)
A_c	FishAssimEfficCarbo	Assimilation efficiency for carbohydrate content of fish feed	-	0.6	(Buschmann, Costa-Pierce et al. 2007)
	SDAProt	Specific dynamic action for digestion of protein	J expended J ⁻¹ protein assimilated across gut wall	0.3	(Stigebrandt 1999)
	SDALipid	Specific dynamic action for digestion of lipid	J expended J ⁻¹ lipid assimilated across gut wall	0.05	(Stigebrandt 1999)
	SDACarbo	Specific dynamic action for digestion of carbohydrate	J expended J ⁻¹ carbohydrate assimilated across gut wall	0.05	(Stigebrandt 1999)
NA	WLs	Scale coefficient in fish weight:length allometry	kg WW mm ^{-1/WLe} fork length	2.84627x10 ⁻⁹	(Petrell and Jones 2000)
NA	WLe	Exponent in fish weight:length allometry	-	3.27	(Petrell and Jones 2000)
α	GWs	Exponent in fish allometric relation between maximal growth rate and live weight	(g WW) ^{1-GWe} d ⁻¹	0.038	(Petrell and Jones 2000)

Coefficient (Stigebrandt 1999)	Coefficient (ROMS code)	Description	Units (Stigebrandt) [ROMS]	Value	Comment
b	GWe	Exponent in fish allometric relation between maximal growth rate and live weight	-	0.667	(Petrell and Jones 2000)
α	resps	Exponent in fish allometric relation between basal respiration rate and live weight	$J (g\ WW)^{-1} d^{-1}$	46.024	(Stigebrandt 1999)
γ	respe	Exponent in fish allometric relation between basal respiration rate and live weight	-	0.74	(Enders and Scruton 2006) but see (Stigebrandt 1999) who suggests 0.8
	Tmptrs	Scale coefficient in exponential relationship governing fish maximal growth and basal respiration	-	1.0	(Stigebrandt 1999). The reference temperature is 0 °C
τ	Tmptre	Exponent coefficient in exponential relationship governing fish maximal growth and basal respiration	°C ⁻¹	0.08	(Stigebrandt 1999)
	SwimCostFrac	Energy expended in swimming relative to basal energy expenditure	$J\ J^{-1}$	1.1	(Petrell and Jones 2000)
	Nresid	Fraction of the protein energy assimilated across the gut wall which is lost as ammonium during protein catabolism	$J\ J^{-1}$	$\frac{E_p^{NO_3} - E_p^{NH_4}}{E_p^{NO_3}}$	(Stigebrandt 1999). Careful reading of (Stigebrandt 1999) reveals that his value for the energy content of protein is based upon complete oxidation. He introduces Nresid=0.15 to account for the energy that is lost because fish oxidize protein only to a NH_4^+ endpoint. That is a little smaller than the value derived from our chosen values for $E_p^{NO_3}$ and $E_p^{NH_4}$

Appendix D Hydrodynamic model vs observations: additional graphs and tables

Tidal height tabulated parameters

Table D-1: Comparison of N2 tidal height parameters for Pelorus 1994–1995 and 1997–1998 tide gauges.
N2 tidal sea level parameters from measurements and model. Here “ratio” means model value divided by observed value and “diffce” means model value minus observed value.

Tide gauge site and deployment	Record length (days)	Amplitude (m)			Phase (°)		
		Obs.	Model	Ratio	Obs.	Model	Diffce
Beatrix East 1994-1995 Deployment 1	54	0.158	0.167	1.06	258.7	264.7	6.0
Pelorus Entrance 1997-1998 Deployment 1	46	0.083	0.077	0.92	201.4	200.8	-0.7
Pelorus Entrance 1997-1998 Deployment 2	69	0.230	0.239	1.04	239.3	242.7	3.4
Pelorus Entrance 1997-1998 Deployment 3	78	0.136	0.147	1.08	276.7	278.7	2.1
Beatrix North 1997-1998 Deployment 1	46	0.089	0.082	0.92	202.0	201.3	-0.7

Table D-2: Comparison of O1 tidal height parameters for Pelorus 1994–1995 and 1997–1998 tide gauges.
As Table D-1 but for the O1 constituent.

Tide gauge site and deployment	Record length (days)	Amplitude (m)			Phase (°)		
		Obs.	Model	Ratio	Obs.	Model	Diffce
Beatrix East 1994-1995 Deployment 1	54	0.019	0.021	1.13	257.1	264.3	7.2
Pelorus Entrance 1997-1998 Deployment 1	46	0.014	0.018	1.35	263.2	236.9	-26.3
Pelorus Entrance 1997-1998 Deployment 2	69	0.020	0.022	1.07	259.0	253.1	-5.9
Pelorus Entrance 1997-1998 Deployment 3	78	0.017	0.016	0.92	236.7	244.3	7.5
Beatrix North 1997-1998 Deployment 1	46	0.013	0.019	1.46	264.1	237.1	-27.0

Tidal velocity graphs

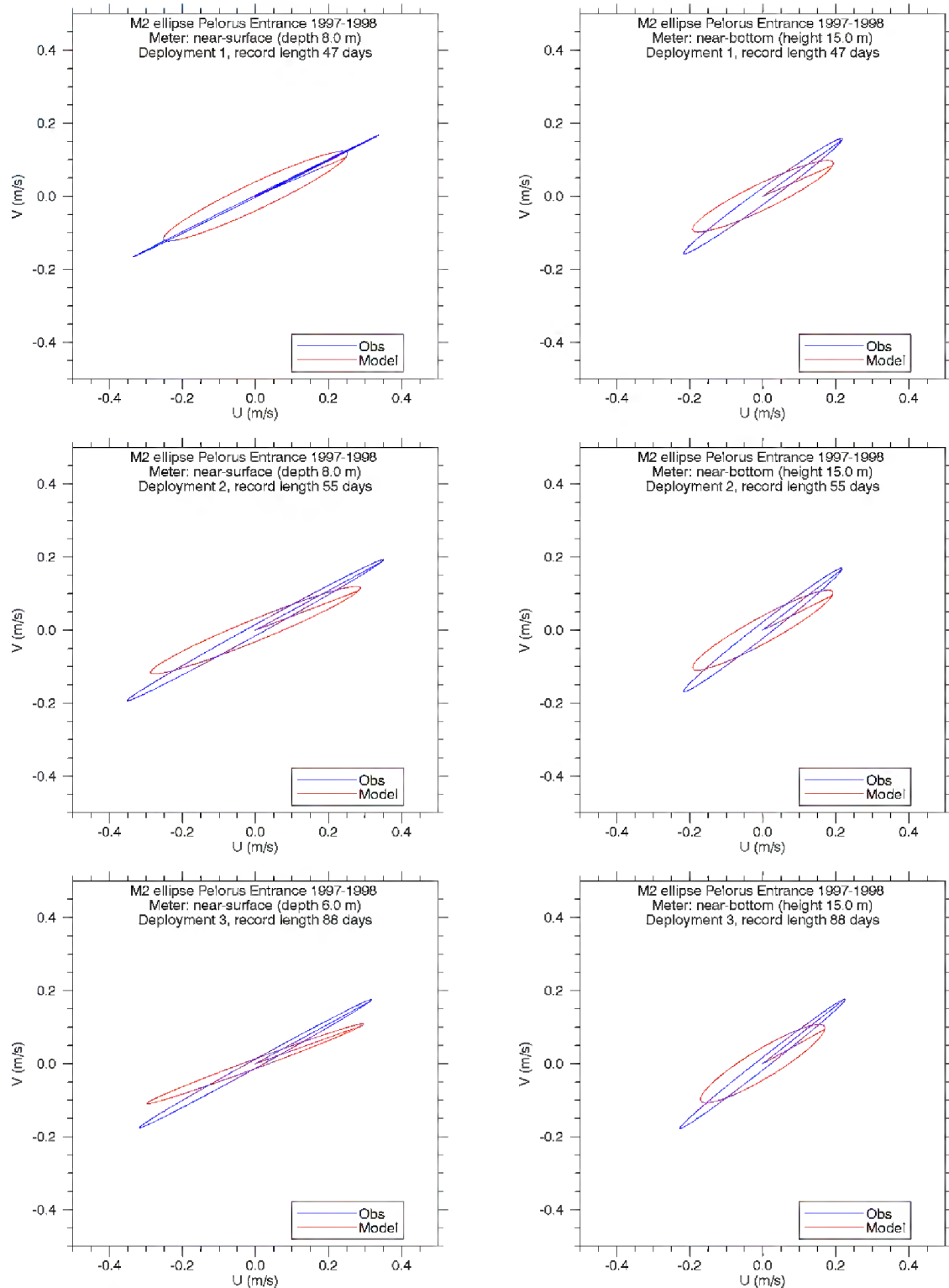


Figure D-1: M2 tidal velocity comparison for Pelorus Entrance 1997–1998. M2 tidal ellipses from current meter (blue) and model (red) at the Pelorus Entrance site, deployments 1 (upper), 2 (middle) and 3 (lower), for near-surface (left) and near-bottom (right) meters. The format of the graphs follows Figure 3-5.

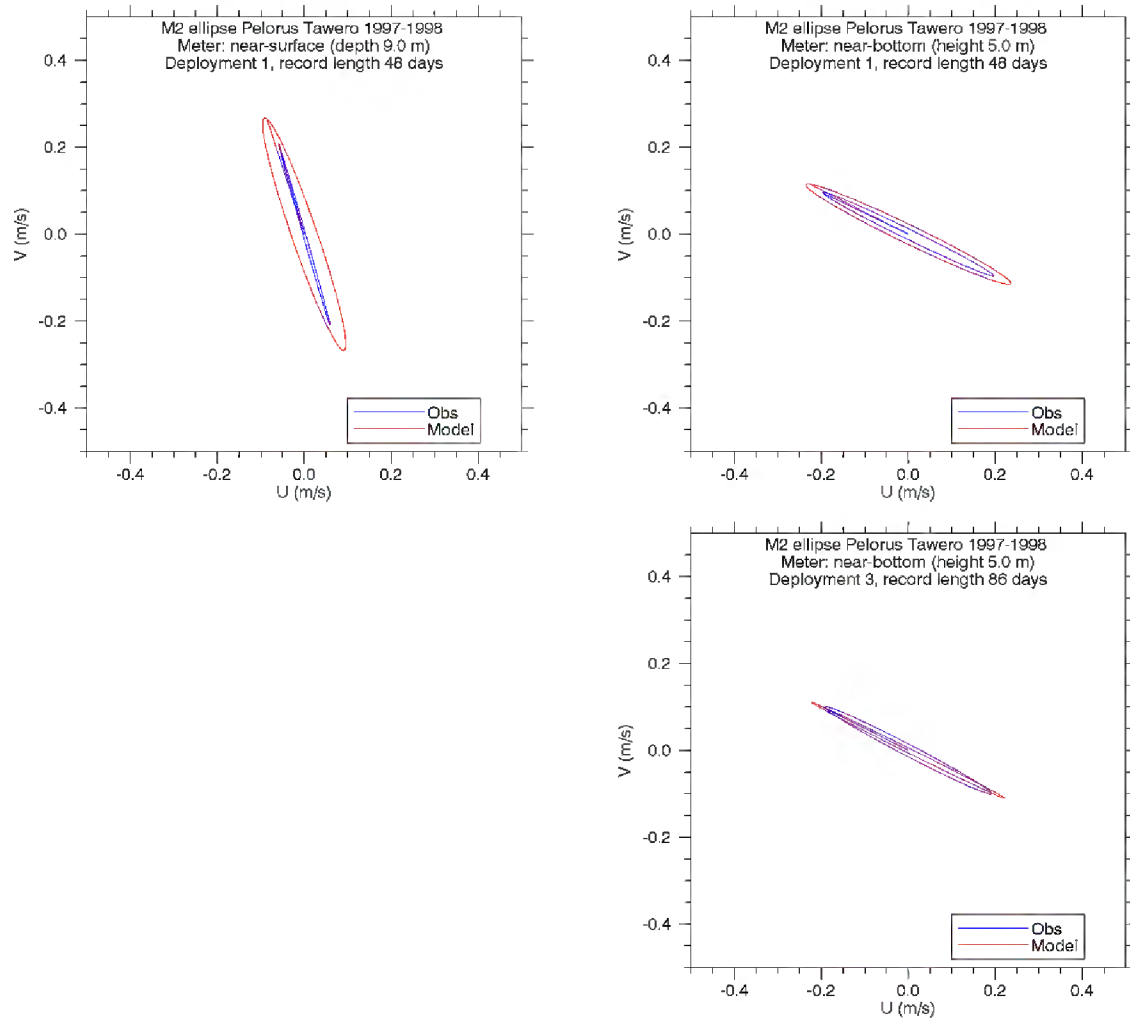


Figure D-2: M2 tidal velocity comparison for Pelorus Tawero 1997–1998. M2 tidal ellipses from current meter (blue) and model (red) at the Pelorus Entrance site, deployments 1 (upper) and 3 (lower), for near-surface (left) and near-bottom (right) meters. The format of the graphs follows Figure 3-5.

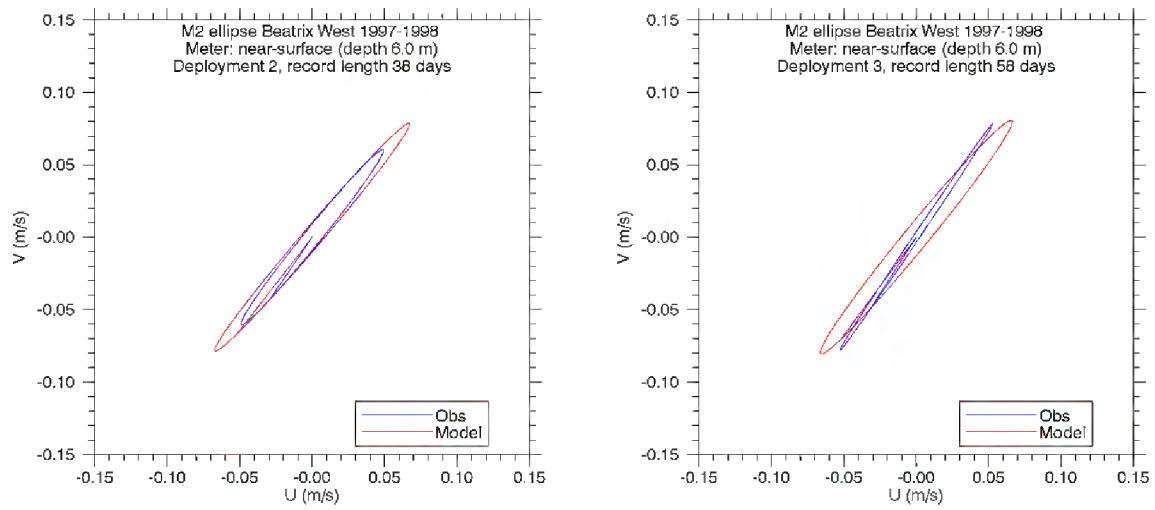


Figure D-3: M2 tidal velocity comparison for Beatrix West 1997–1998. M2 tidal ellipses from current meter (blue) and model (red) at the Beatrix West site, near-surface meters, for deployments 2 (left) and 3 (right). The format of the graphs follows Figure 3-5.

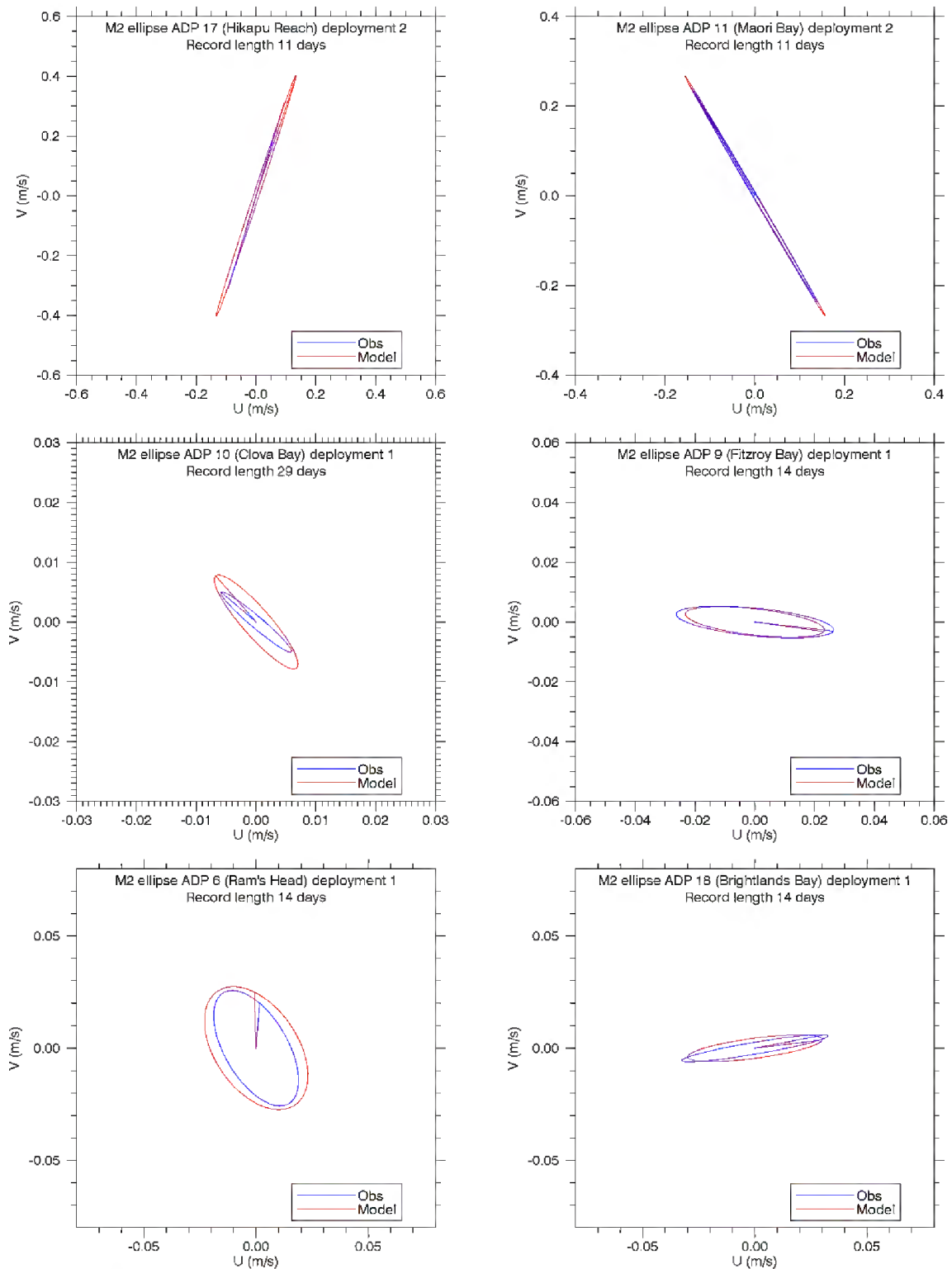


Figure D-4: M2 tidal velocity comparison for 6 FRIA 2005 ADCPs. M2 tidal ellipses of depth-averaged current from ADCP (blue) and model (red). Sites and deployments as indicated. The remaining 3 sites are shown in Figure D-5.

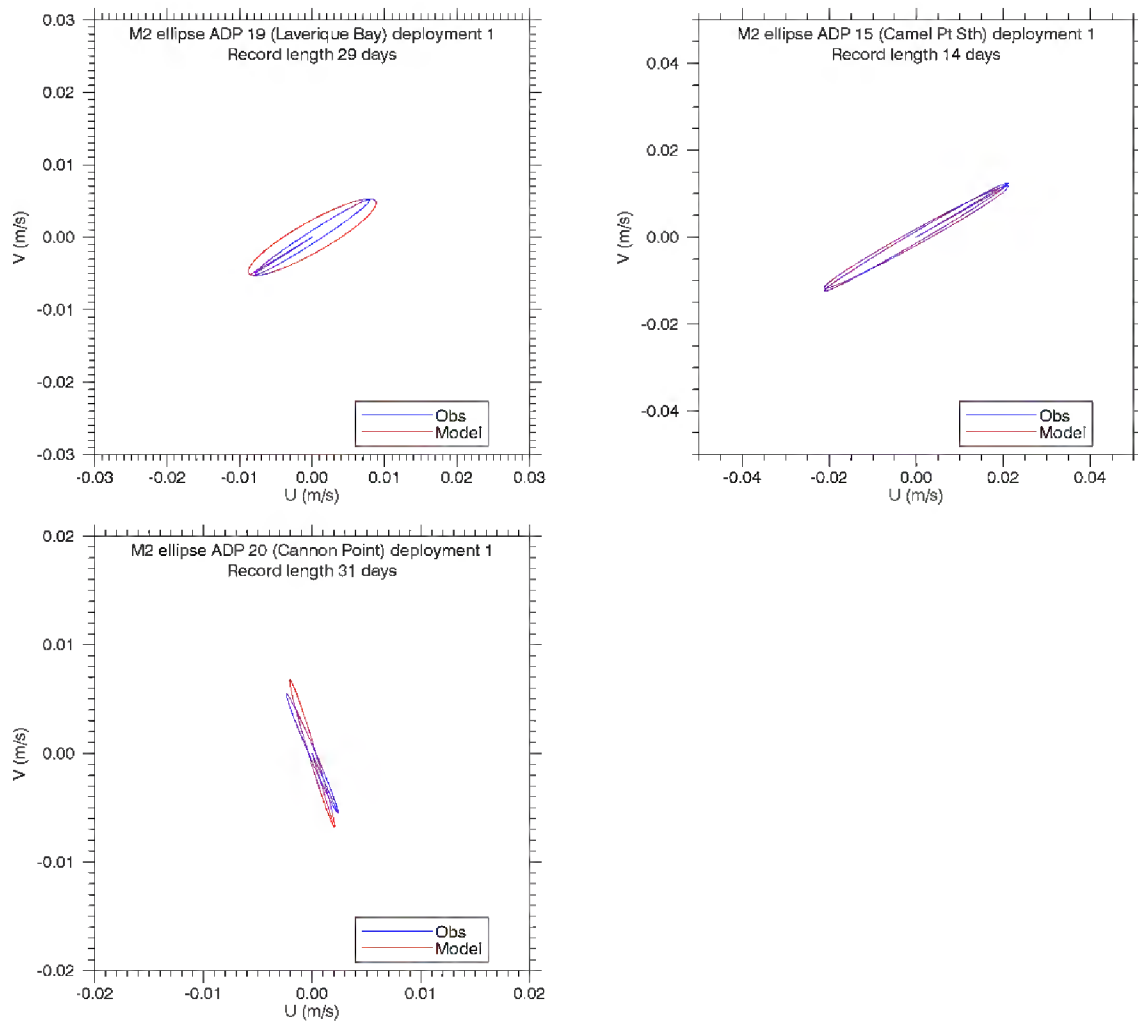


Figure D-5: M2 tidal velocity comparison for 3 FRIA 2005 ADCPs. M2 tidal ellipses of depth-averaged current from ADCP (blue) and model (red). Sites and deployments as indicated. The previous 6 sites are shown in Figure D-4.

Subtidal velocity scatter plots

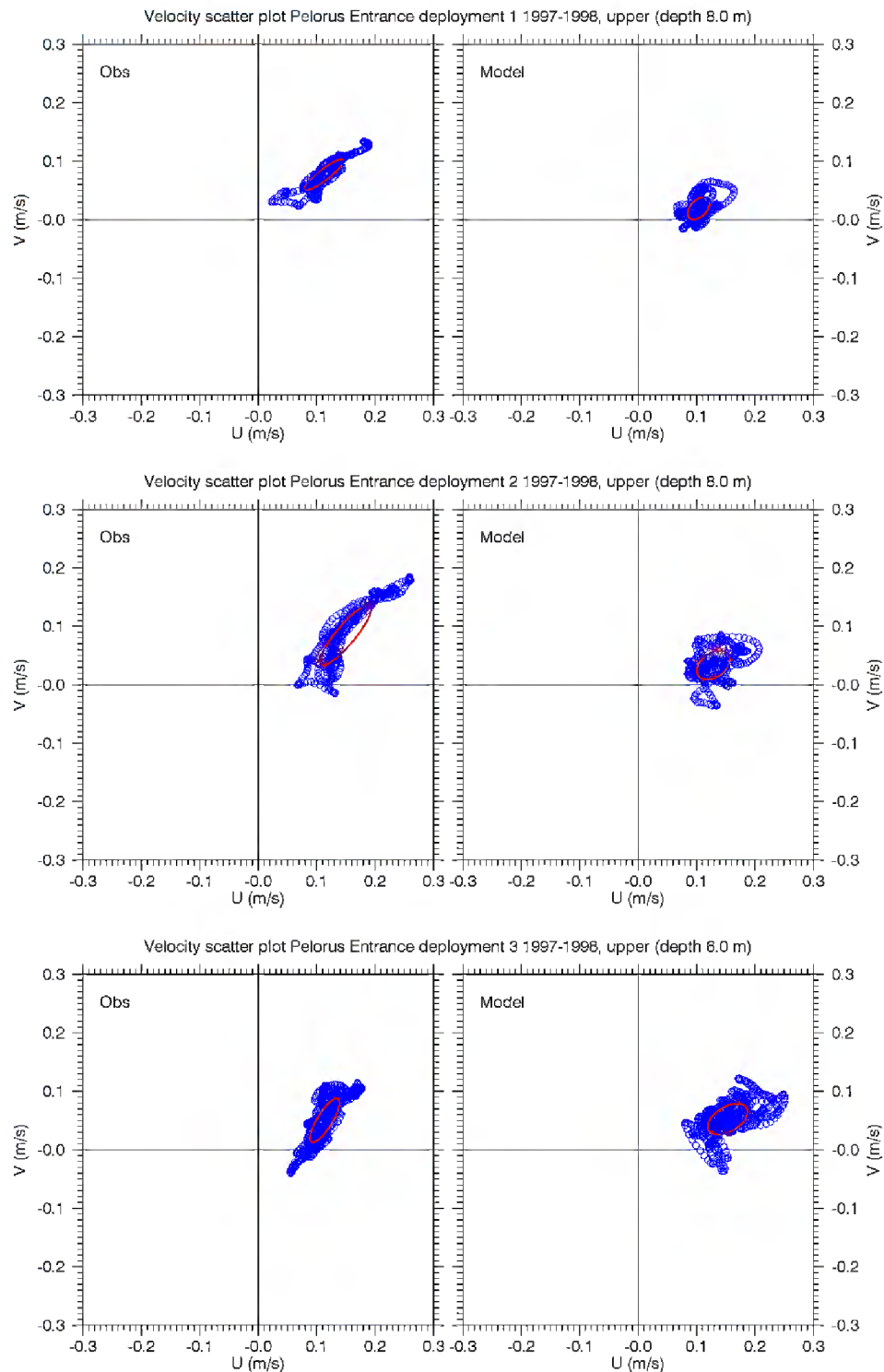


Figure D-6: Subtidal velocity comparison for Pelorus Entrance upper current meter. Subtidal velocity scatter plots from current meter (left) and model (right) at the Pelorus Entrance upper site, deployments 1 (upper), 2 (middle) and 3 (lower). The format of the graphs follows Figure 3-6.

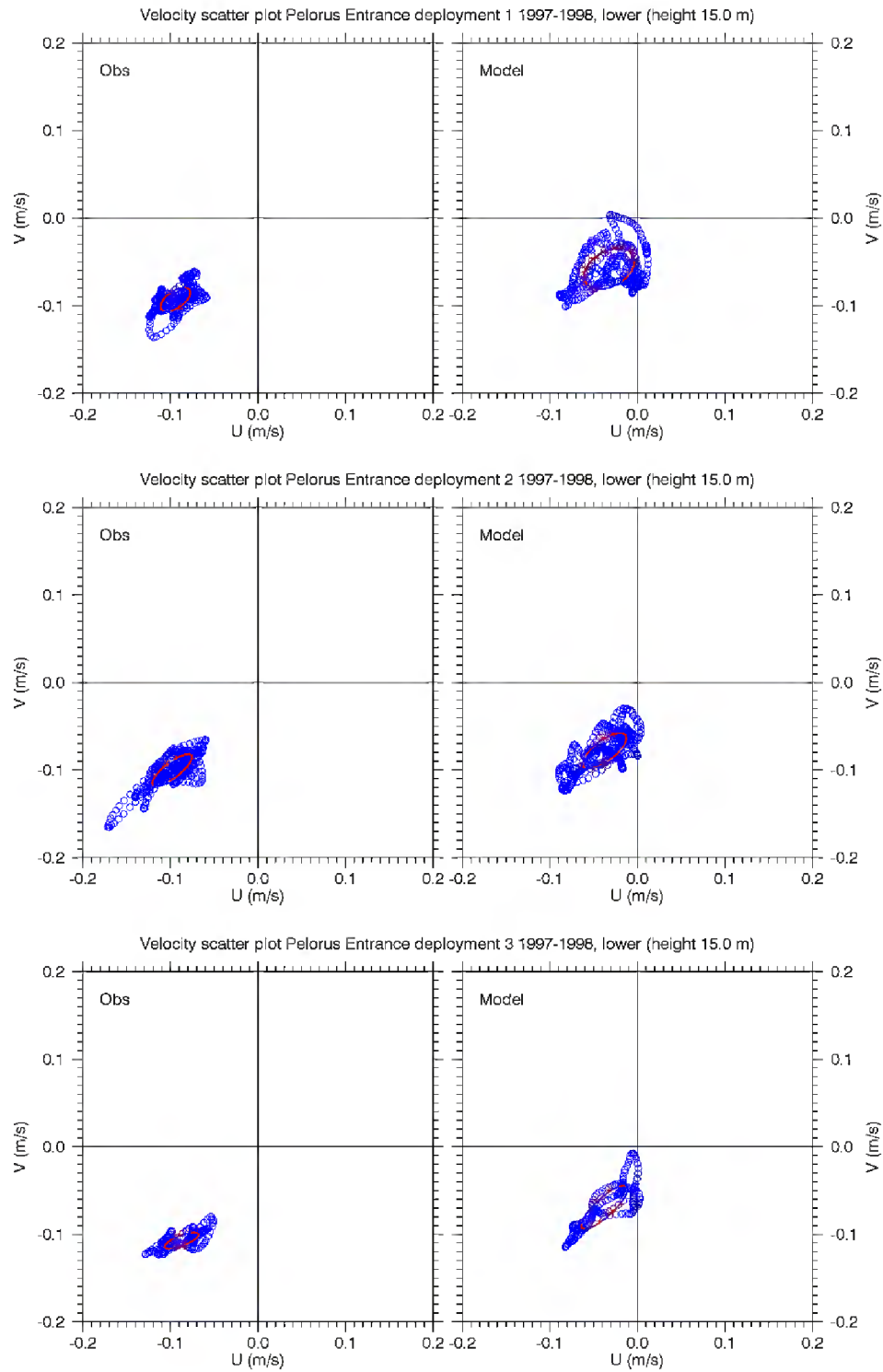


Figure D-7: Subtidal velocity comparison for Pelorus Entrance lower current meter. As Figure D-6 but for the lower current meter.

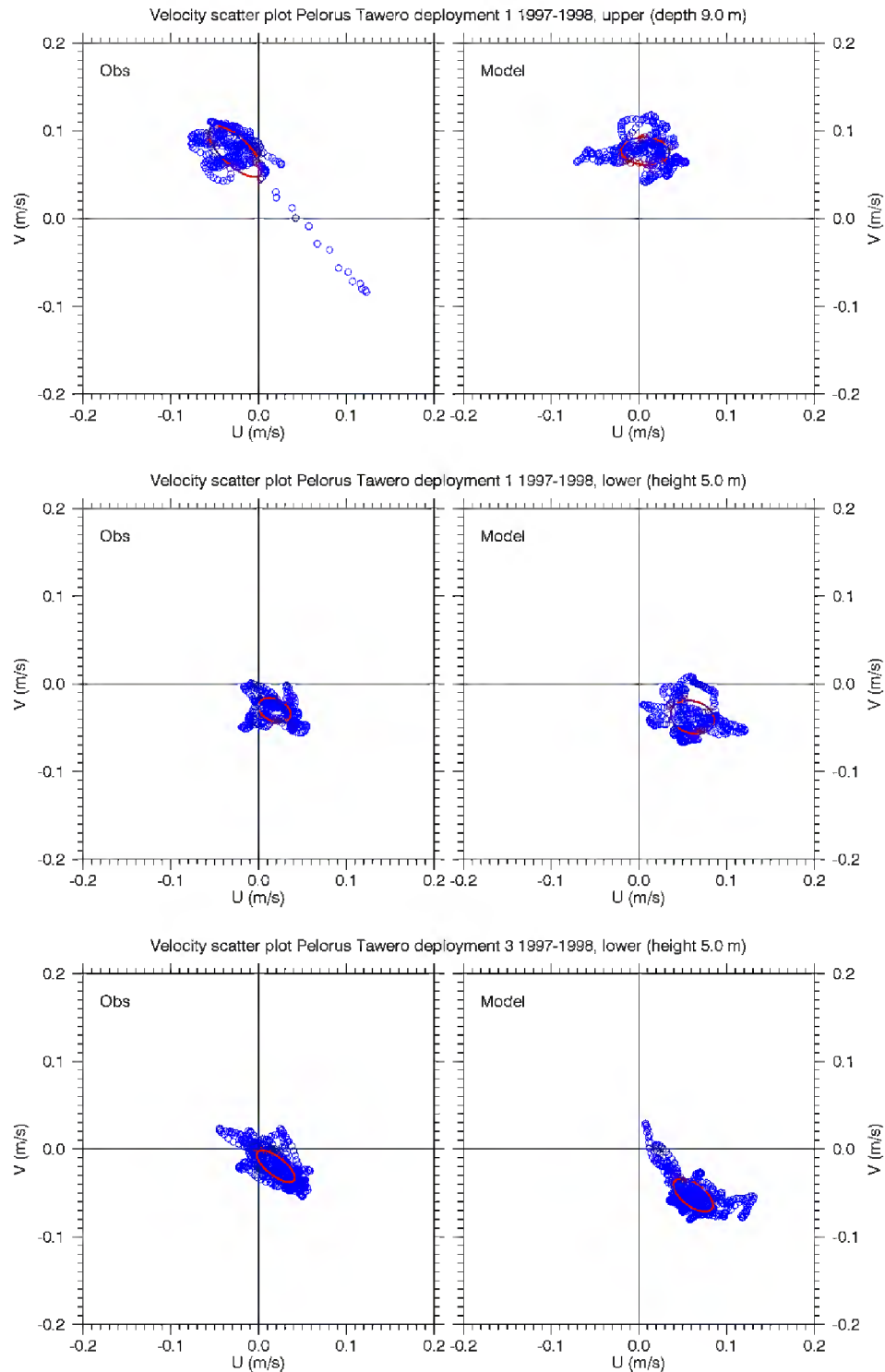


Figure D-8: Subtidal velocity comparison for Pelorus Tawero current meters. Subtidal velocity scatter plots from current meter (left) and model (right) at the Pelorus Tawero site, from top: deployment 1 upper; deployment 1 lower; deployment 3 lower.. The format of the graphs follows Figure 3-6.

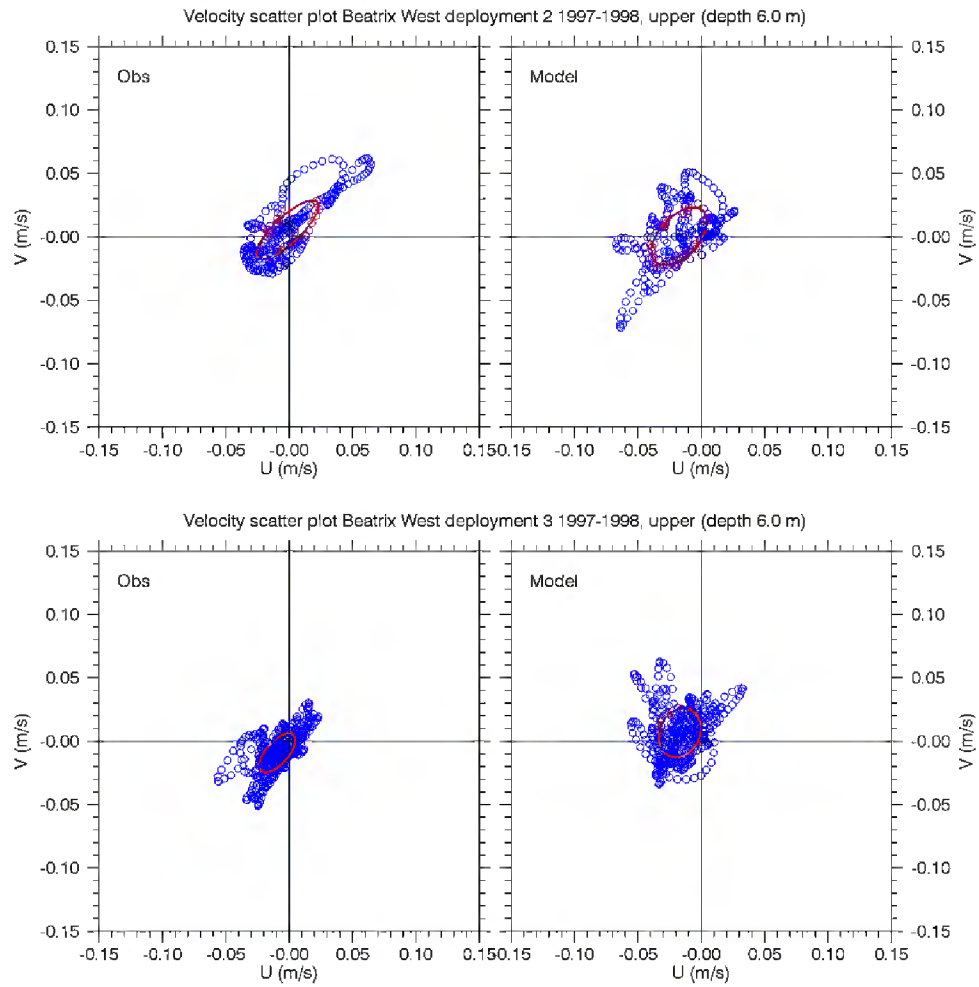


Figure D-9: Subtidal velocity comparison for Beatrix West current meters. Subtidal velocity scatter plots from current meter (left) and model (right) at the Beatrix West site, from top: deployment 2 upper; deployment 3 upper. The format of the graphs follows Figure 3-6.

Tidal velocity tabulated parameters

Table D-3: Comparison of M2 tidal ellipse parameters for Pelorus 1994–1995 and 1997–1998 current meters. M2 tidal ellipse parameters from current meter measurements and model. Here “ratio” means model value divided by measured value and “diffce” means model value minus measured value.

Current meter Site/Level/Deployment	Record length (days)	Semi-major axis (m/s)			Eccentricity			Inclination (°T)			Phase (°)		
		Meas.	Model	Ratio	Meas.	Model	Diffce	Meas.	Model	Diffce	Meas.	Model	Diffce
Pelorus Entrance near-surface (depth 8.0 m) deployment 1	47	0.376	0.278	0.74	0.01	0.13	0.12	63.6	64.8	1.2	9.2	14.6	5.4
Pelorus Entrance near-surface (depth 8.0 m) deployment 2	55	0.401	0.310	0.77	0.03	0.09	0.06	61.2	67.8	6.6	11.6	13.6	2.0
Pelorus Entrance near-surface (depth 6.0 m) deployment 3	88	0.364	0.317	0.87	0.03	0.04	0.01	61.1	69.9	8.7	11.4	12.2	0.8
Pelorus Entrance near-bottom (height 15.0 m) deployment 1	47	0.268	0.214	0.80	-0.07	0.16	0.23	54.1	64.2	10.1	352.1	12.2	20.2
Pelorus Entrance near-bottom (height 15.0 m) deployment 2	55	0.275	0.219	0.80	-0.06	0.16	0.22	52.1	61.3	9.1	354.0	14.2	20.2
Pelorus Entrance near-bottom (height 15.0 m) deployment 3	88	0.288	0.198	0.69	-0.05	0.20	0.24	52.0	58.9	6.9	5.2	10.8	5.6
Pelorus Tawero near-surface (depth 9.0 m) deployment 1	48	0.217	0.283	1.30	-0.01	0.10	0.12	164.2	161.1	-3.1	359.3	9.3	10.0
Pelorus Tawero near-bottom (height 5.0 m) deployment 1	48	0.219	0.262	1.20	-0.05	0.08	0.13	116.0	115.8	-0.2	3.3	13.8	10.5
Pelorus Tawero near-bottom (height 5.0 m) deployment 3	86	0.216	0.248	1.15	-0.06	0.02	0.08	117.5	116.4	-1.1	10.7	15.2	4.5
Beatrix West near-surface (depth 6.0 m) deployment 2	38	0.078	0.103	1.33	0.07	0.06	-0.01	39.0	40.5	1.5	132.0	144.7	12.7
Beatrix West near-surface (depth 6.0 m) deployment 3	58	0.094	0.104	1.10	0.02	0.08	0.06	34.0	39.3	5.3	134.7	147.3	12.6

Table D-4: Comparison of S2 tidal ellipse parameters for Pelorus 1994–1995 and 1997–1998 current meters. As Table D-3 but for the S2 constituent.

Current meter Site/Level/Deployment	Record length (days)	Semi-major axis (m/s)			Eccentricity			Inclination (°T)			Phase (°)		
		Meas.	Model	Ratio	Meas.	Model	Diffce	Meas.	Model	Diffce	Meas.	Model	Diffce
Pelorus Entrance near-surface (depth 8.0 m) deployment 1	47	0.126	0.106	0.85	0.05	0.16	0.12	63.2	59.6	-3.6	92.2	84.7	-7.6
Pelorus Entrance near-surface (depth 8.0 m) deployment 2	55	0.161	0.136	0.84	0.06	0.10	0.04	56.5	58.3	1.8	60.7	51.7	-8.9
Pelorus Entrance near-surface (depth 6.0 m) deployment 3	88	0.110	0.093	0.85	0.07	0.23	0.16	55.8	59.7	3.9	82.3	67.8	-14.5
Pelorus Entrance near-bottom (height 15.0 m) deployment 1	47	0.136	0.114	0.84	-0.09	0.00	0.09	54.3	55.3	1.0	67.1	95.8	28.6
Pelorus Entrance near-bottom (height 15.0 m) deployment 2	55	0.165	0.133	0.81	-0.09	0.04	0.13	53.4	53.1	-0.3	44.6	61.7	17.1
Pelorus Entrance near-bottom (height 15.0 m) deployment 3	88	0.132	0.095	0.72	-0.11	0.02	0.12	54.9	50.1	-4.8	49.9	75.2	25.3
Pelorus Tawero near-surface (depth 9.0 m) deployment 1	48	0.092	0.120	1.30	-0.21	-0.23	-0.02	1.6	175.1	-6.5	86.9	85.4	-1.6
Pelorus Tawero near-bottom (height 5.0 m) deployment 1	48	0.109	0.147	1.35	-0.03	0.15	0.18	108.0	101.2	-6.8	85.6	92.6	7.0
Pelorus Tawero near-bottom (height 5.0 m) deployment 3	86	0.089	0.112	1.27	0.02	0.16	0.14	113.2	110.1	-3.0	79.5	73.6	-5.9
Beatrix West near-surface (depth 6.0 m) deployment 2	38	0.028	0.050	1.74	0.04	0.10	0.06	37.2	41.0	3.8	186.8	193.6	6.7
Beatrix West near-surface (depth 6.0 m) deployment 3	58	0.031	0.032	1.06	0.14	0.19	0.06	29.0	24.3	-4.8	202.9	217.3	14.4

Table D-5: Comparison of M2 tidal ellipse parameters for FRIA 2005 ADCPs. M2 tidal ellipse parameters from ADCP measurements and model. Here “ratio” means model value divided by measured value and “diffce” means model value minus measured value.

ADCP Site/Deployment	Record length (days)	Semi-major axis (m/s)			Eccentricity			Inclination (°T)			Phase (°)		
		Meas.	Model	Ratio	Meas.	Model	Diffce	Meas.	Model	Diffce	Meas.	Model	Diffce
Site 17 deployment 2	11	0.326	0.424	1.30	0.00	0.02	0.02	16.8	18.3	1.5	11.7	11.8	0.2
Site 11 deployment 2	11	0.274	0.309	1.13	0.01	-0.01	-0.03	149.5	149.7	0.2	19.9	16.1	-3.8
Site 10 deployment 1	29	0.008	0.010	1.33	0.13	-0.23	-0.36	130.3	138.7	8.4	0.1	359.7	-0.4
Site 9 deployment 1	14	0.026	0.023	0.89	-0.18	-0.19	-0.01	95.7	96.7	1.0	174.8	173.6	-1.3
Site 6 deployment 1	14	0.029	0.031	1.07	-0.49	-0.60	-0.11	149.5	145.7	-3.8	305.0	313.7	8.7
Site 18 deployment 1	14	0.033	0.030	0.91	0.08	0.15	0.07	80.4	82.4	2.0	344.7	3.7	19.1
Site 19 deployment 1	29	0.009	0.010	1.07	-0.08	0.21	0.29	57.1	60.5	3.4	173.6	171.9	-1.7
Site 15 deployment 1	14	0.025	0.024	0.98	0.05	0.07	0.02	59.6	61.1	1.5	359.8	350.8	-9.0
Site 20 deployment 1	31	0.006	0.007	1.19	-0.05	0.05	0.1	156.6	163.2	6.6	180	185.9	5.9

Subtidal velocity tabulated parameters

Table D-6: Sub-tidal velocity comparison. Sub-tidal mean and variance ellipse parameters from ADCP measurements and model, and temporal correlations between measured and modelled time series. Here “ratio” means model value divided by measured value and “diffce” means model value minus measured value.

ADCP Site & Deployment	Mean magnitude (m/s)			Mean direction (°T)			Semi-major axis (m/s)			Eccentricity			Inclination (°T)			Correlation	
	Meas.	Model	Diffce	Meas.	Model	Diffce	Meas.	Model	Ratio	Meas.	Model	Diffce	Meas.	Model	Diffce	Along-chann.	Across-chann.
Pelorus Entrance 1997-1998 upper deployment 1	0.136	0.105	-0.031	55.7	79.5	23.8	0.043	0.023	0.54	0.22	0.61	0.39	52.60	43.50	-9.00	0.60	0.38
Pelorus Entrance 1997-1998 upper deployment 2	0.172	0.133	-0.038	60.4	75.5	15.0	0.067	0.032	0.48	0.25	0.65	0.40	40.00	55.00	15.10	0.53	0.64
Pelorus Entrance 1997-1998 upper deployment 3	0.124	0.162	0.038	66.3	71.2	4.9	0.044	0.038	0.86	0.30	0.56	0.27	31.60	58.90	27.40	0.50	0.33
Pelorus Entrance 1997-1998 lower deployment 1	0.133	0.066	-0.066	225.3	208.9	-16.4	0.020	0.031	1.59	0.50	0.65	0.15	55.20	60.20	5.10	0.39	0.51
Pelorus Entrance 1997-1998 lower deployment 2	0.140	0.087	-0.054	224.3	205.6	-18.7	0.028	0.030	1.07	0.32	0.40	0.08	52.10	53.30	1.20	0.71	0.39
Pelorus Entrance 1997-1998 lower deployment 3	0.138	0.079	-0.060	219.1	208.5	-10.6	0.021	0.035	1.68	0.29	0.30	0.01	68.70	47.20	-21.40	0.80	-0.04
Pelorus Tawero 1997-1998 upper deployment 1	0.081	0.077	-0.004	340.9	5.7	24.8	0.039	0.028	0.71	0.40	0.57	0.17	133.10	93.20	-39.90	0.40	0.40
Pelorus Tawero 1997-1998 lower deployment 1	0.035	0.072	0.037	148.8	121.9	-26.9	0.019	0.026	1.32	0.65	0.71	0.07	114.20	106.80	-7.50	0.44	-0.41
Pelorus Tawero 1997-1998 lower deployment 3	0.028	0.081	0.053	135.0	130.3	-4.7	0.026	0.027	1.01	0.40	0.49	0.09	128.10	126.90	-1.20	0.08	-0.24
Beatrix West 1997-1998 upper deployment 2	0.006	0.017	0.012	345.8	271.1	-74.7	0.032	0.028	0.86	0.32	0.56	0.24	46.70	43.00	-3.80	0.45	0.30
Beatrix West 1997-1998 upper deployment 3	0.013	0.018	0.005	225.2	291.0	65.8	0.019	0.020	1.05	0.46	0.79	0.33	41.70	24.50	-17.20	0.29	0.21

Appendix E Time-averaged water-quality properties in the bottom-most layer of the water-column

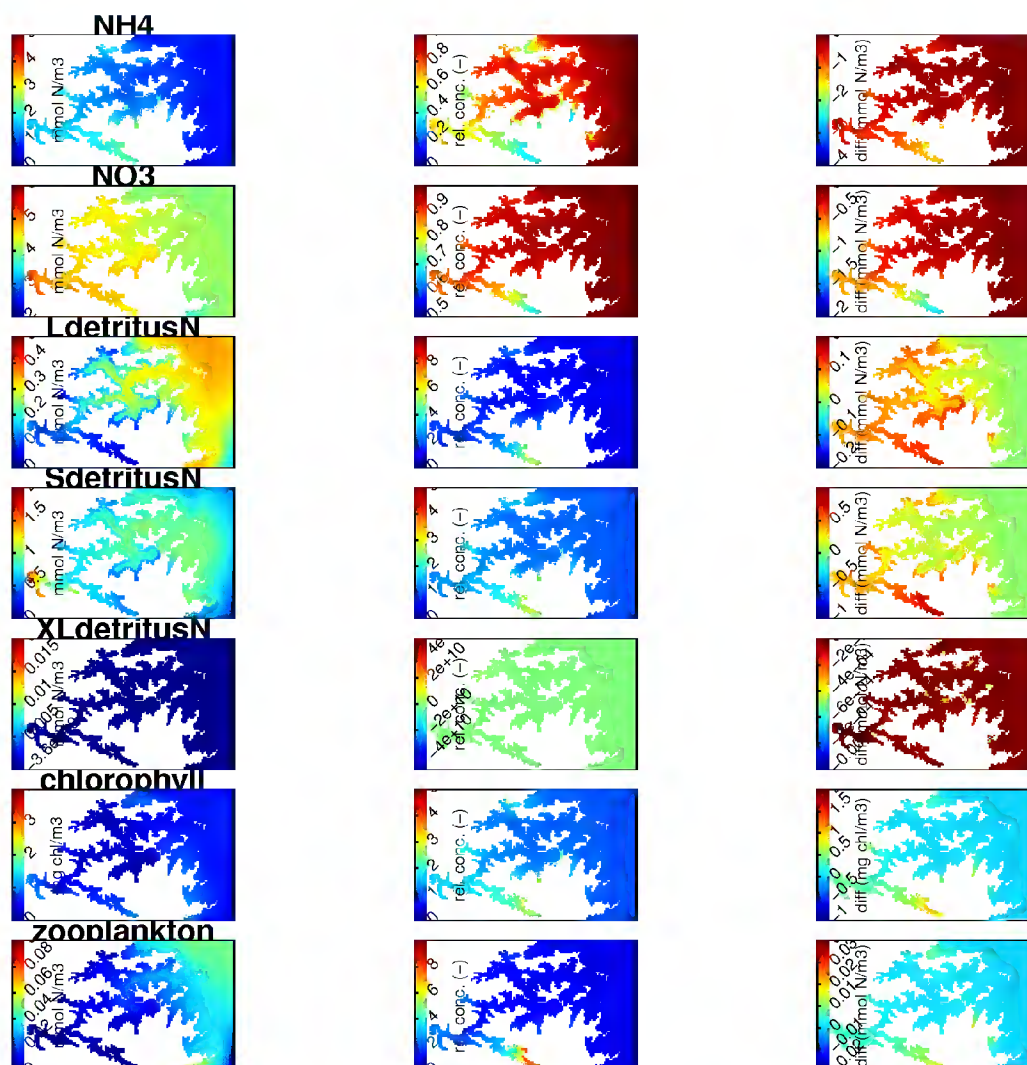


Figure E-1: Comparison of winter time-averaged concentrations in the EM-EF-WD and NM-EF-WD scenarios in the bottom-most layer of the water-column. The left-hand panel illustrates the time-average in the surface-most layer for the reference scenario (EM-EF-WD). The central panel illustrates the time-averaged relative concentration (alternative scenario relative to reference). The right hand column illustrates the time-averaged concentration difference (alternative scenario - reference scenario). These simulations were made on a 400 m horizontal resolution grid.

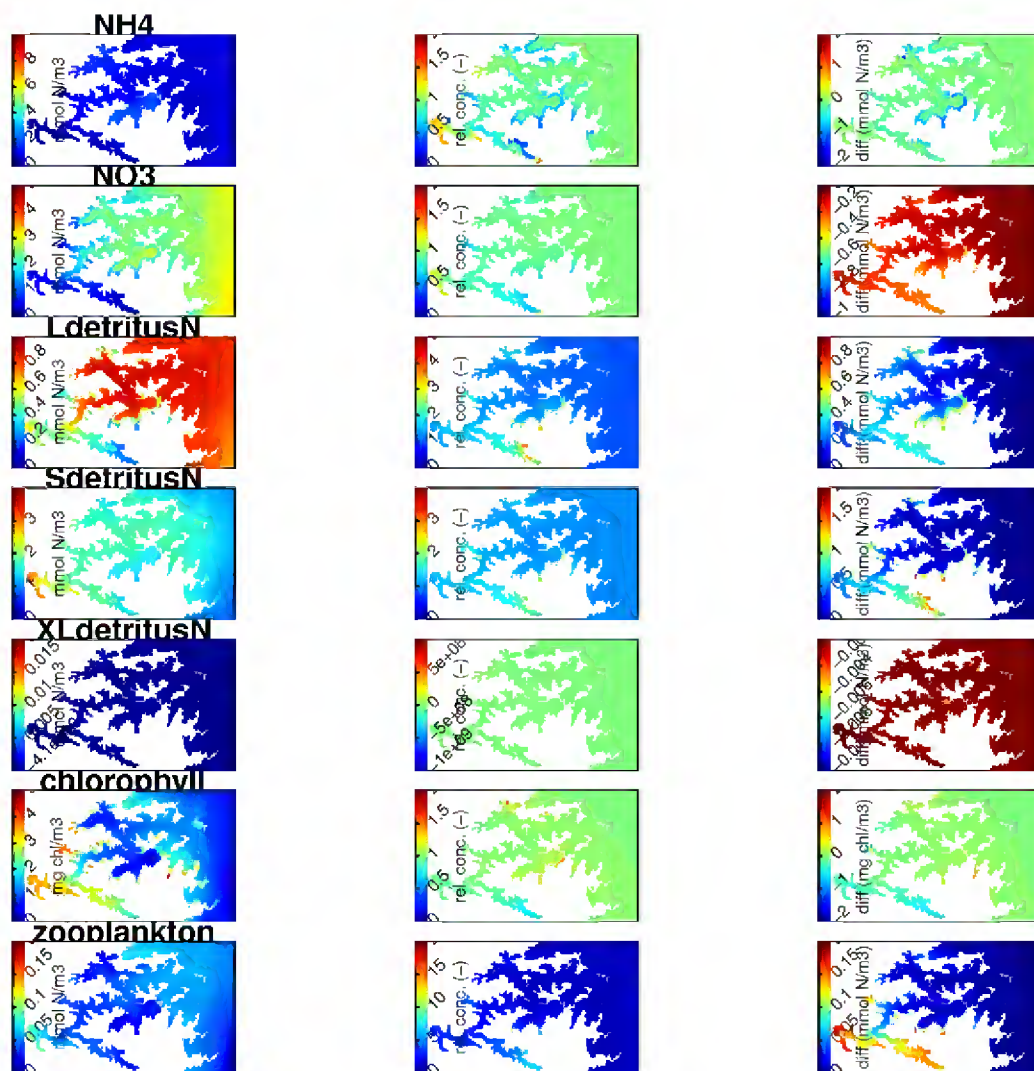


Figure E-2: Comparison of summer time-averaged concentrations in the EM-EF-WD and NM-EF-WD scenarios in the bottom-most layer of the water-column. Refer to the caption of Figure E-1 for further details. These simulations were made on a 400 m horizontal resolution grid.

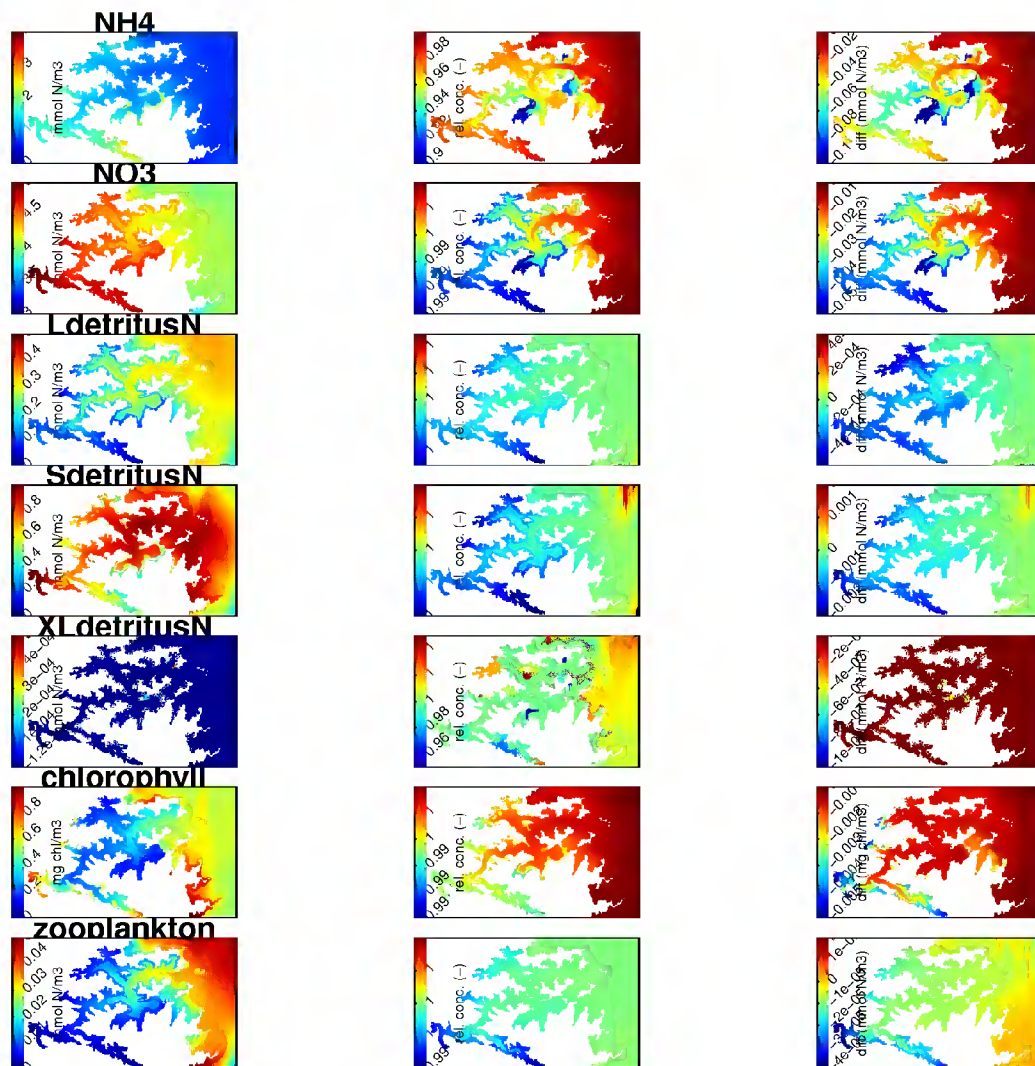


Figure E-3: Comparison of winter time-averaged concentrations in the EM-EF-WD and EM-NF-WD scenarios in the bottom-most layer of the water-column. Refer to the caption of Figure E-1 for further details.

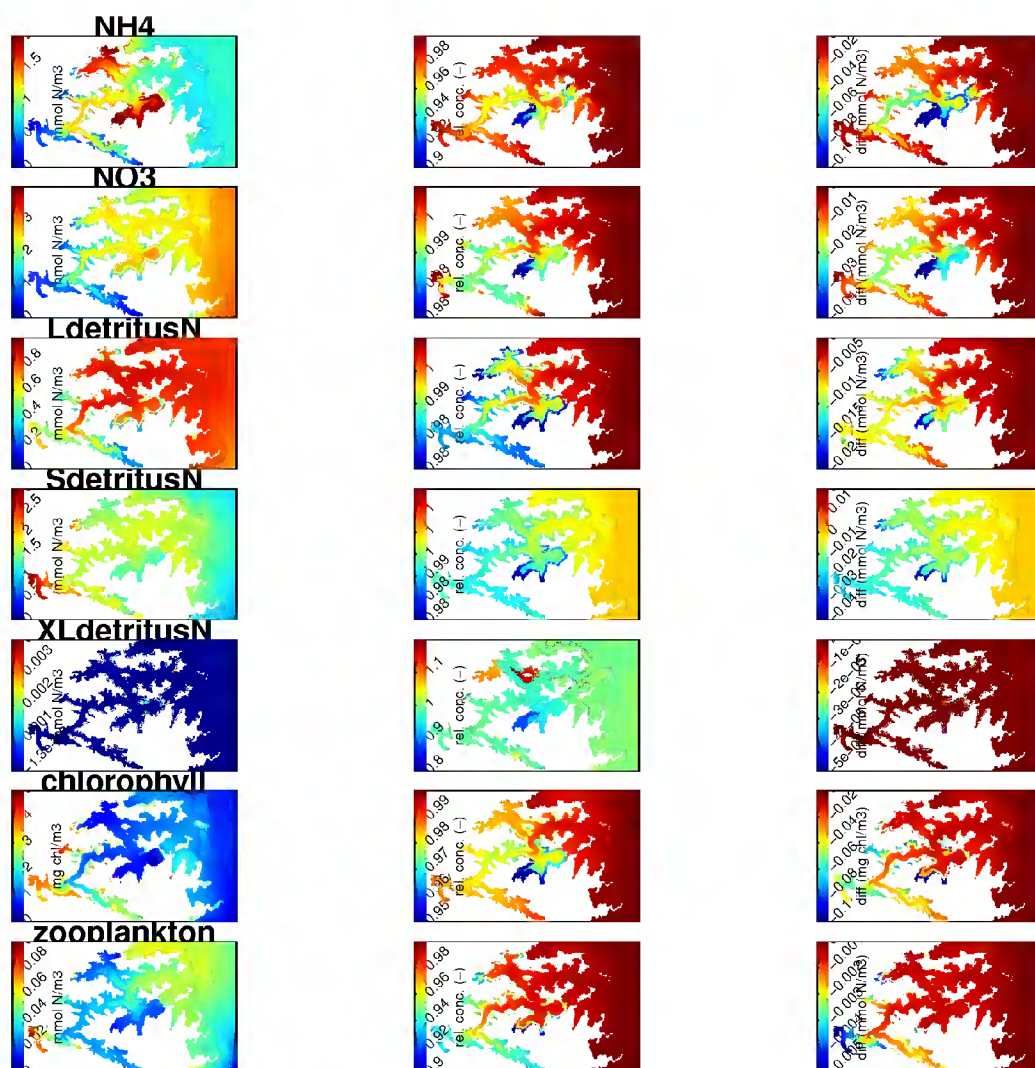


Figure E-4: Comparison of summer time-averaged concentrations in the EM-EF-WD and EM-NF-WD scenarios in the bottom-most layer of the water-column. Refer to the caption of Figure E-1 for further details.

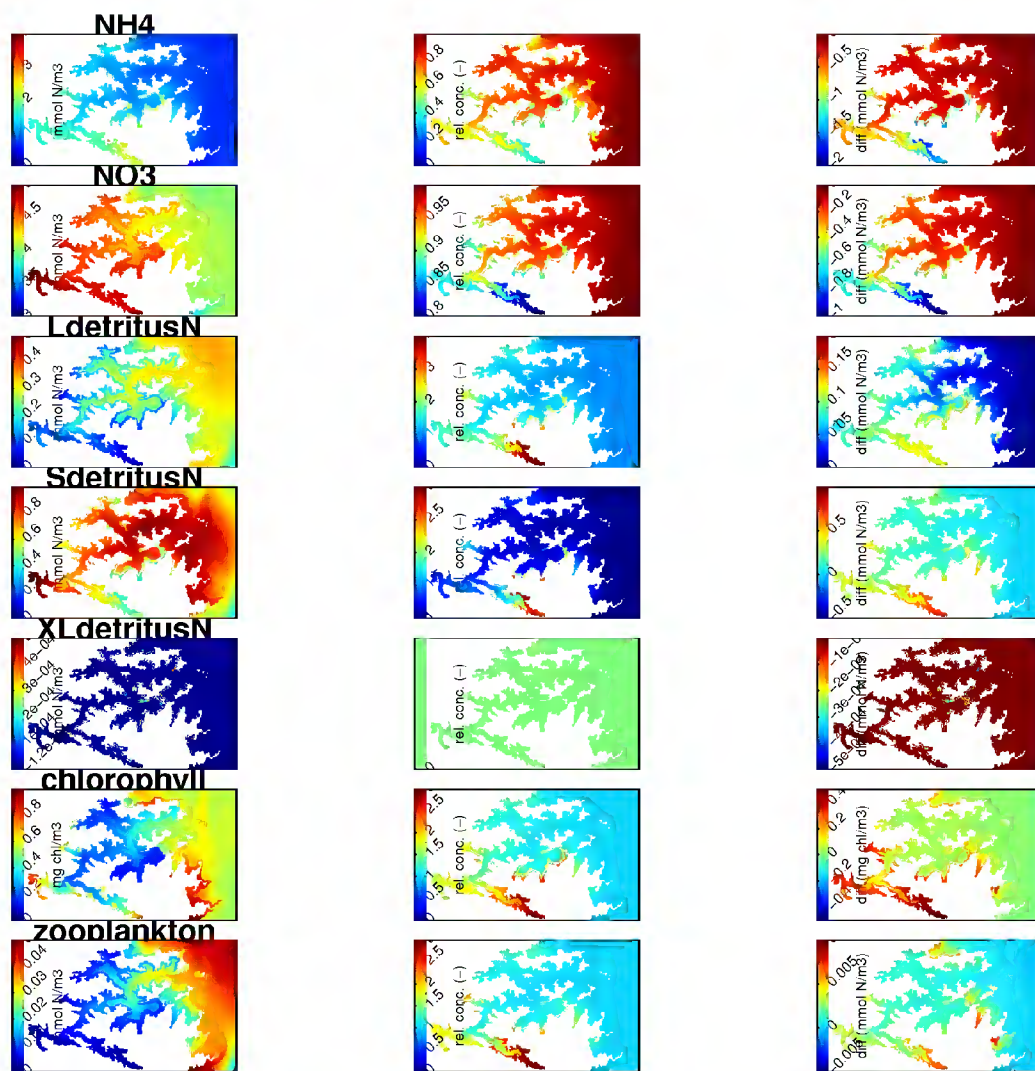


Figure E-5: Comparison of winter time-averaged concentrations in the EM-EF-WD and NM-NF-WD scenarios in the bottom-most layer of the water-column. Refer to the caption of Figure E-1 for further details.

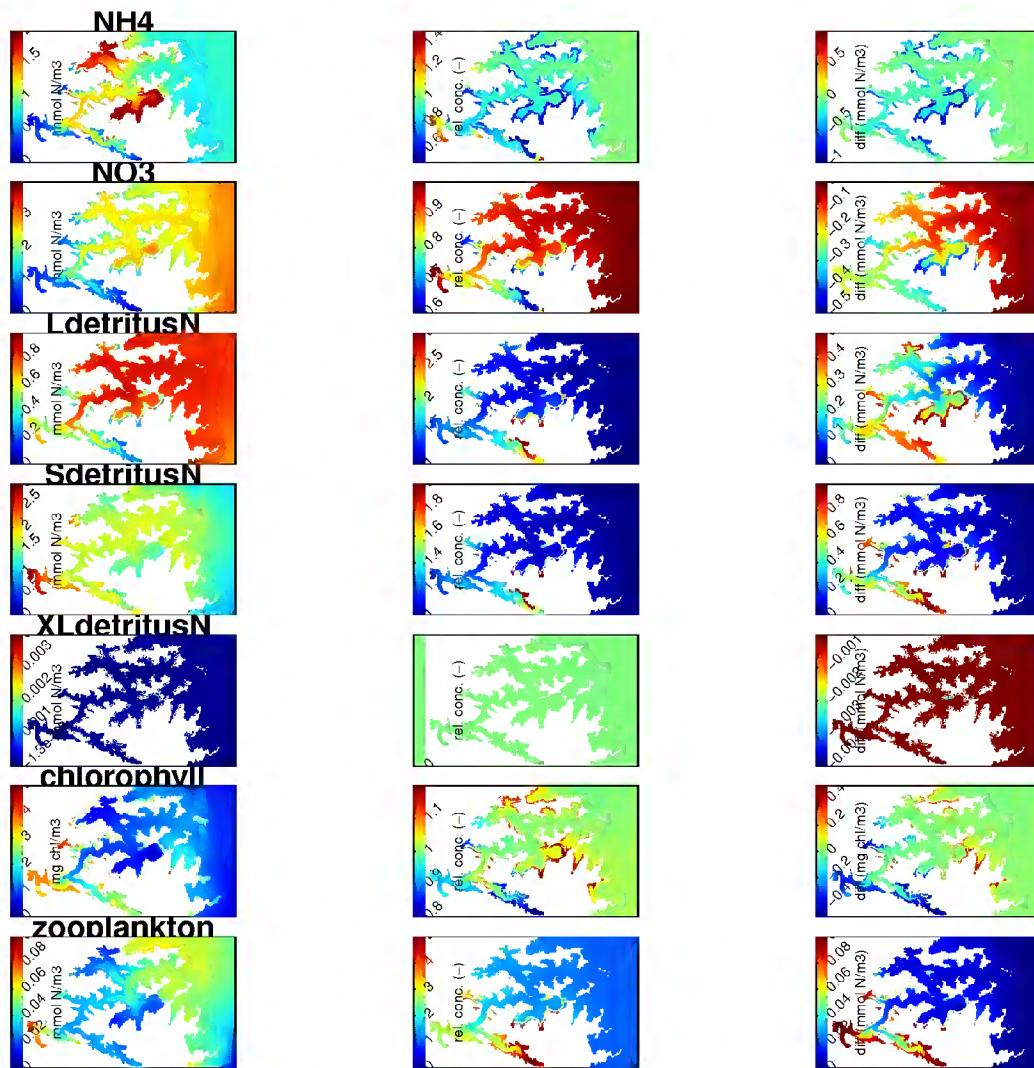


Figure E-6: Comparison of summer time-averaged concentrations in the EM-EF-WD and NM-NF-WD scenarios in the bottom-most layer of the water-column. Refer to the caption of Figure E-1 for further details.

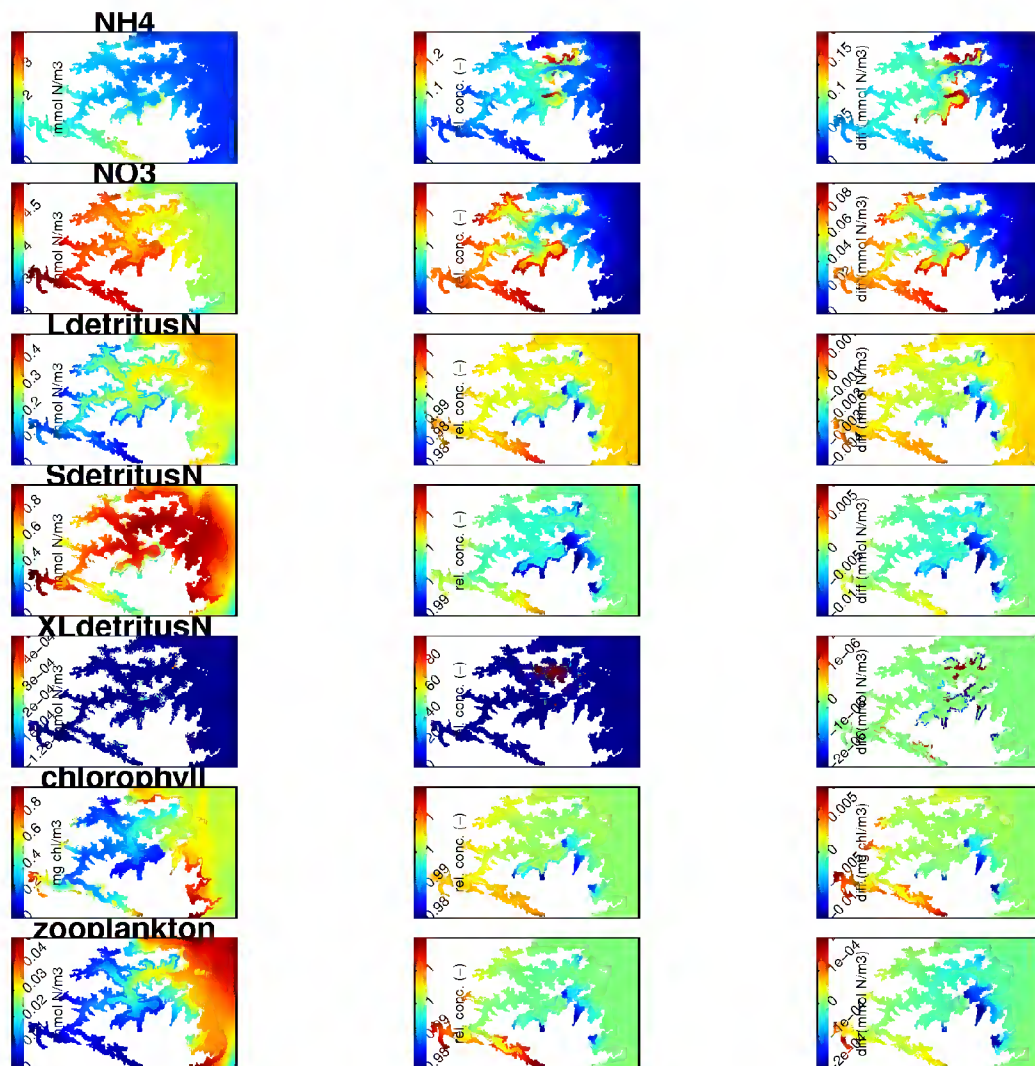


Figure E-7: Comparison of winter time-averaged concentrations in the EM-EF-WD and AM-AF-WD scenarios in the bottom-most layer of the water-column. Refer to the caption of Figure E-1 for further details.

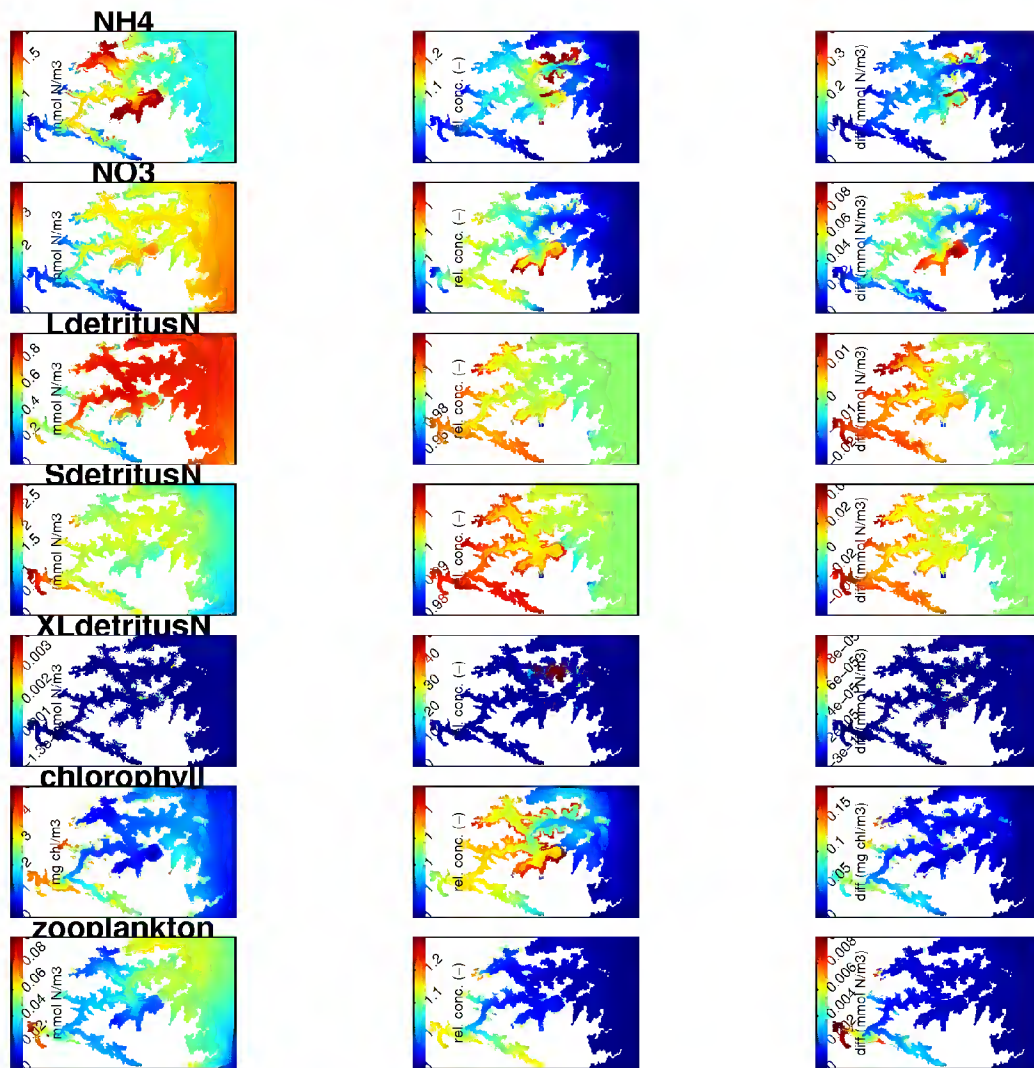


Figure E-8: Comparison of summer time-averaged concentrations in the EM-EF-WD and AM-AF-WD scenarios in the bottom-most layer of the water-column. Refer to the caption of Figure E-1 for further details.

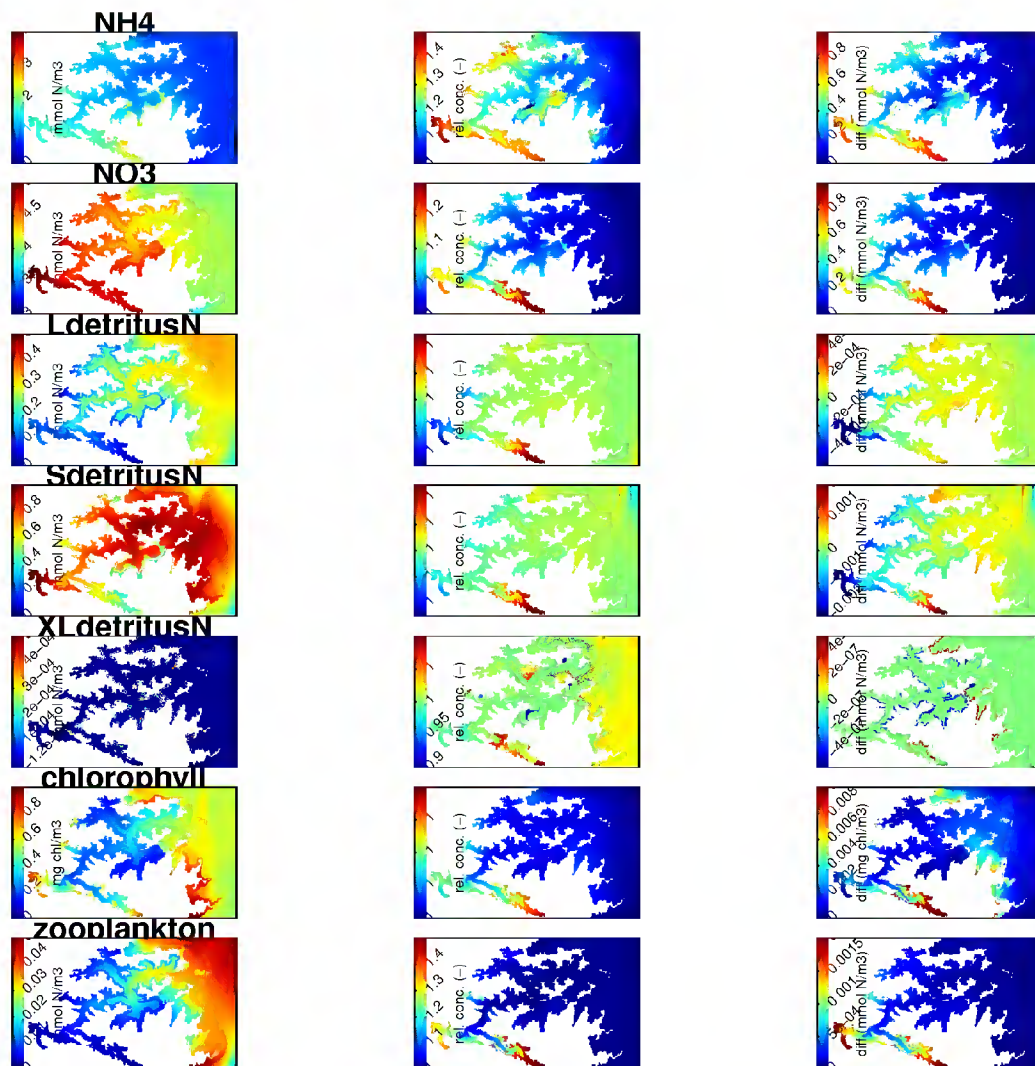


Figure E-9: Comparison of winter time-averaged concentrations in the EM-NF-WD and EM-NF-ND scenarios in the bottom-most layer of the water-column. Note that the reference scenario (EM-NF-WD) differs from that used in most comparisons. Refer to the caption of Figure E-1 for further details

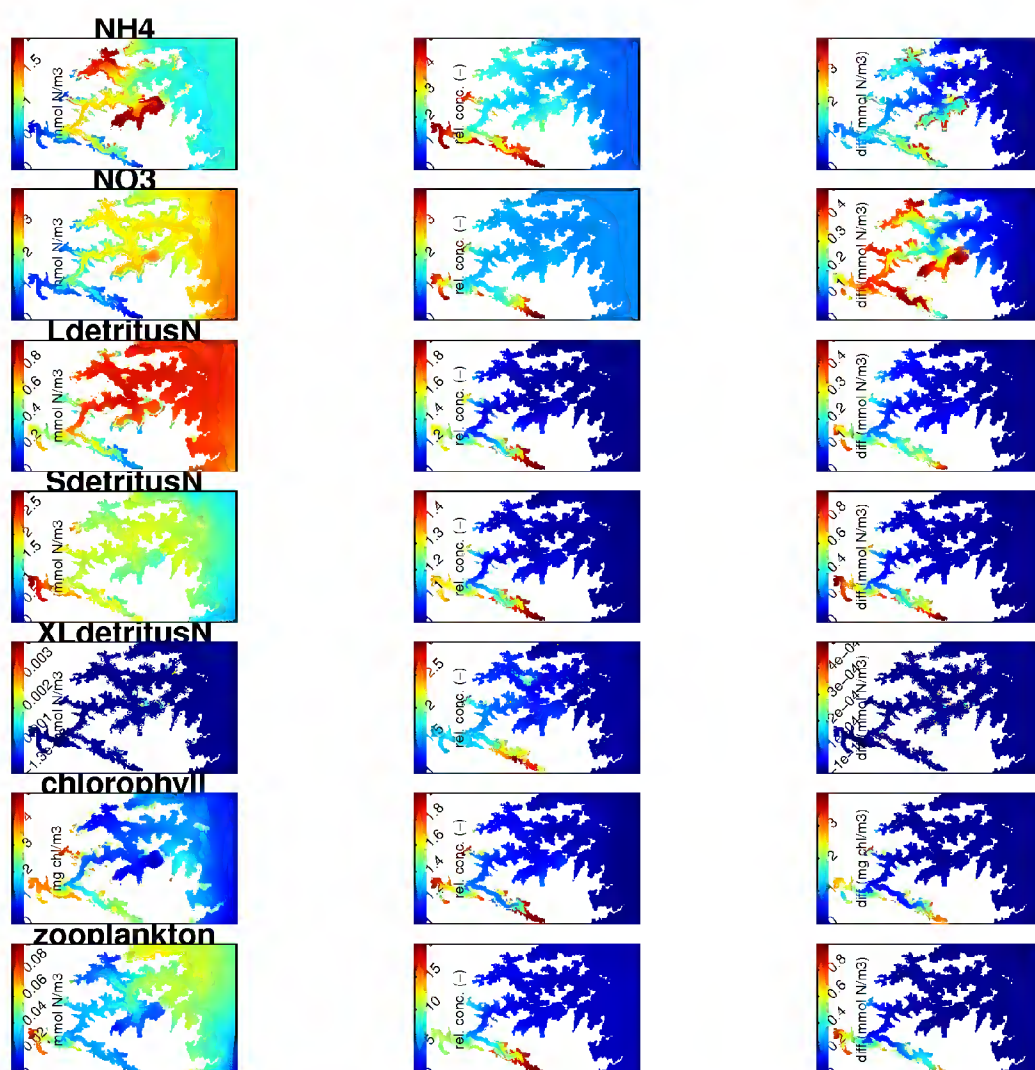


Figure E-10: Comparison of summer time-averaged concentrations in the EM-NF-WD and EM-NF-ND scenarios in the bottom-most layer of the water-column. Note that the reference scenario (EM-NF-WD) differs from that used in most comparisons. Refer to the caption of Figure E-1 for further details

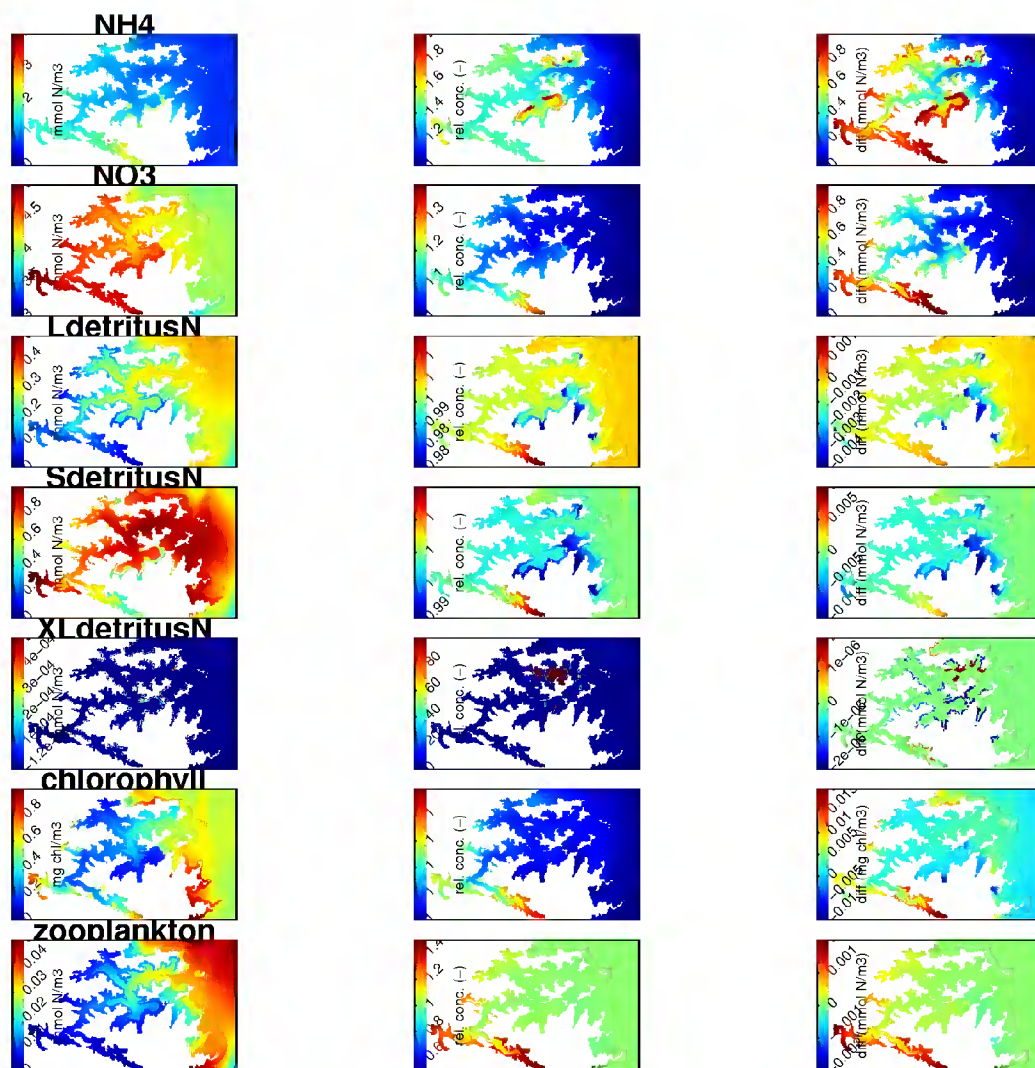


Figure E-11: Comparison of winter time-averaged concentrations in the EM-EF-WD and AM-AF-ND scenarios in the bottom-most layer of the water-column. Refer to the caption of Figure E-1 for further details.

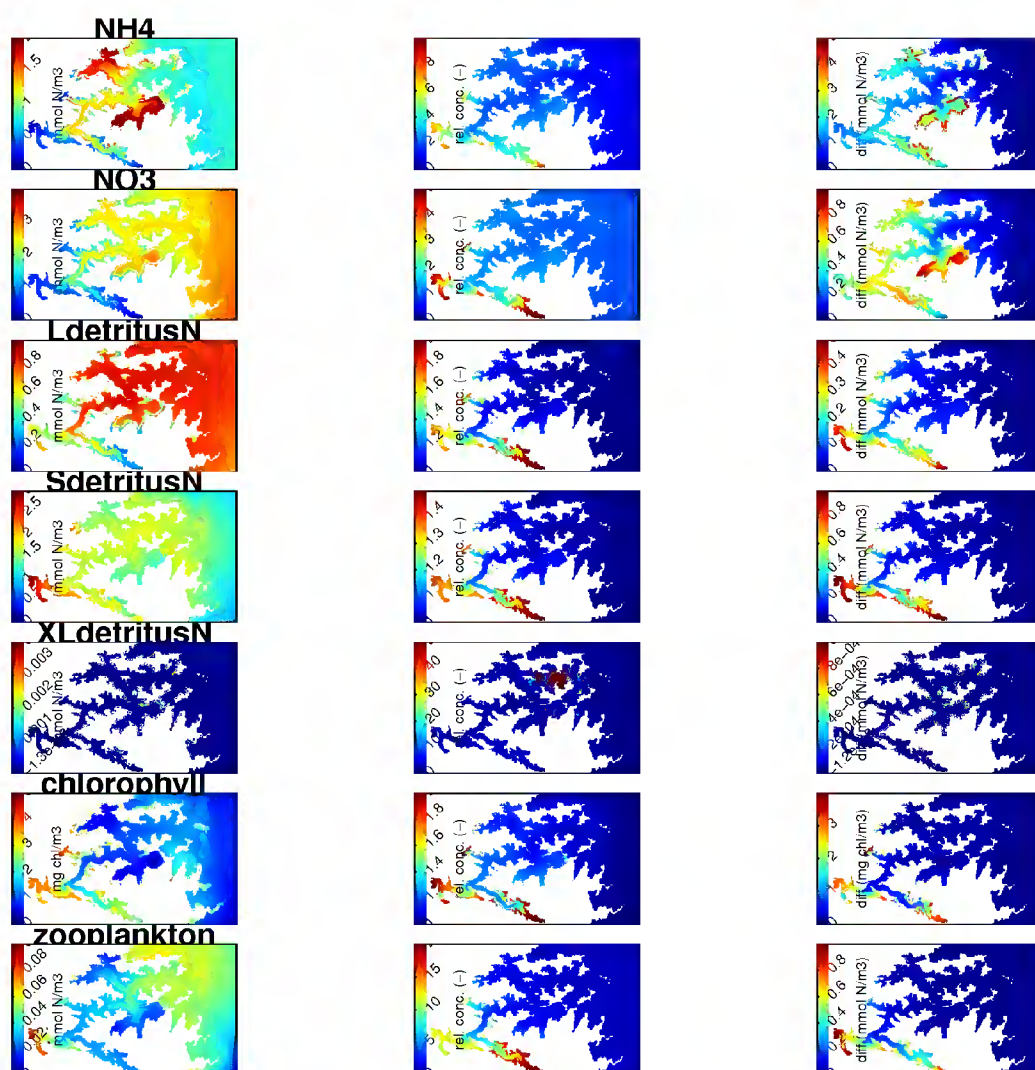


Figure E-12: Comparison of winter time-averaged concentrations in the EM-EF-WD and AM-AF-ND scenarios in the bottom-most layer of the water-column. Refer to the caption of Figure E-1 for further details.

Attachment 9

Dr ST Mead CV



Shaw Mead. PhD

Nationality: New Zealand

Date of Birth: 3rd February 1967

Profession: Environmental Scientist/Consultant

Contact Details: Ph: +64 21 423 224 E: s.mead@ecoast.co.nz

PRESENT POSITIONS:

- Managing Director/Environmental Scientist, eCoast Ltd
- Director, Eco Surf Viti Ltd
- Lecturer and Research Provider, Unitec.

SPECIALISATION:

Coastal oceanography/engineering, beach processes, coastal hazards/SLR, climate change adaptation and resilience strategies, ecology (marine, freshwater, and terrestrial), coastal structure design and impact assessment, surf science, hydrodynamic and sediment transport numerical modelling, aquaculture, environmental impact assessment, expert witness/reviewer.

YEARS OF EXPERIENCE: 26

KEY QUALIFICATIONS:

- PhD in Coastal Oceanography & Numerical Modelling, University of Waikato, New Zealand (1996-2000)
- MSc (Hons) in Environmental Science & Marine Ecology/Aquaculture, University of Auckland, New Zealand (1994-96)
- BSc in Marine Biology & Botany, University of Auckland, New Zealand (1991-93)

PROFESSIONAL MEMBERSHIPS:

- New Zealand Coastal Society (ENZ)
- New Zealand Association of Impact Assessment
- Technical Advisor for the Surfbreak Protection Society (NZ) and Save the Waves Coalition
- Editorial Board for the Journal of Coastal Conservation, Planning and Management
- New Zealand Fiji Business Council
- Registered Environmental Impact Assessment (EIA) consultant in Fiji (coastal processes, coastal engineering, marine ecology, numerical modelling)

PERSONAL STATEMENT:

Dr Mead's background in coastal oceanography and marine ecology, specialising in hydrodynamic and sediment transport numerical modelling, coastal processes, coastal structures/processes interactions, marine ecology and aquaculture, allows him to effectively bridge the multi-disciplinary gap between physical processes and marine ecological impacts. His PhD thesis in physical oceanography is based on a series of peer-reviewed papers that together with more than 30 popular articles, have presented novel techniques to record the shape of the seabed and surfing breaks, specify the breaking intensity of waves and to break-down surf breaks into their morphological components using numerical modelling.

Dr Mead's research and consulting have led to major advances in our knowledge of offshore reefs for the development of multiple-use structures (coastal protection, amenities such as surfing, wind-surfing, diving, fishing, and ecological enhancement), and have incorporated numerical modelling of waves, currents and sediment transport to develop the designs and assess the impacts of coastal structures over a large range of spatial and temporal scales. Dr Mead is a world-leader in the discipline of surf science and multi-purpose reef design and research, enabling the incorporation of high-quality surfing reefs into multi-purpose coastal structures. This work has also been applied to the development of recreational wave-pool designs and patents, and advising on wave generation techniques for a range of applications (e.g. for water scenes in the King Kong movie, the development of a multi-wave surf pool facility, etc.). He has also applied surf science to the protection and understanding of natural surfing breaks, was instrumental in the incorporation of New Zealand's nationally significant surfing breaks into the New Zealand Coastal Policy Statement (NZCPS, 2010), and was recently part of a government-funded research team developing the world's first set of management and protection guidelines for surf breaks.

Commercially, Dr Mead has been involved in development of beach management and coastal remediation/protection strategies, assessments of coastal hazards, marina and beach design, ecological and physical effects of marine construction, dredging, oil industry and aquaculture ventures, ecological and physical effects of subdivisions and outfalls, development of climate change resilience strategies to sea level rise, and the management and protection of surfing breaks.

Practical Experience

Dr Mead is currently an environmental scientist and Managing Director at eCoast, which is a marine consulting and research organization, focussed on applying up to date knowledge on physical and biological processes in a holistic approach to coastal, estuarine and freshwater management. Dr Mead has over 26 years' experience in marine research and consulting, has published 60 peer-reviewed publications, 2 chapters in Marine and Coastal Resource Management: Principles and Practise ('Beach Management', and 'Surf Science and Multi-Purpose Reefs'), and has solely or jointly produced over 450 technical reports pertaining to coastal management/hazards, coastal structures, erosion control, beach remediation and management strategies, surf break protection and management, marina and beach design, marine, freshwater and terrestrial ecology, coastal oceanography and aquaculture.

Dr Mead has undertaken more than 2500 consulting and research SCUBA dives around the coast of New Zealand, the Pacific Islands, Indian Ocean Islands, South Africa, Europe, Indonesia and North America, and is affiliated to the New Zealand Coastal Society (ENZ). Dr Mead is also experienced at presenting and providing expert witness evidence at resource consent Hearings and in Environment and High Court, EPA hearings, as well as public meetings and seminars. To keep up to date with the latest advances in coastal science and numerical modelling, Dr Mead regularly attends national and international conferences such as the ICCE, the Australasian Coasts and Ports and NZCS (ENZ).

Dr Mead has led or been involved in a range of projects related to all aspects of coastal management and development including design and impact studies of coastal structures, beaches, marinas and multi-purpose reefs (coastal protection, amenity and ecological enhancement) (in NZ, UK, Australia, US, Fiji, Costa Rica, Mexico, South Africa, New Caledonia, Malaysia), coastal process investigations to identify the causes and remedies for coastal erosion (in NZ, Australia, UK, South Africa, Indian Ocean Islands), port expansion (breakwaters and directed wave-driven currents to reduce maintenance dredging), oil field development (NZ and Australia), habitat enhancement (in NZ), artificial reef designs for enhancement of fisheries and tourism (in NZ, Persian Gulf, Fiji), surf break impacts and management, and beach management and climate change resilience strategies.

These projects have included field data acquisition, data analysis, design, impact assessment, public consultation and application for permits. Clients have included central and local government authorities, private and public corporations, international engineering and management companies, private developers, hotels and resorts and aid providers (NZODA, ADB, SPC, UNDP, GIZ). Dr Mead took the role of the Coastal Engineer/Scientist for the Tongan and Marshall Islands components of the Pacific Community's Global Climate Change Alliance: Pacific Small Island States (GCCA: PSIS), a programme which received the 2019 Energy Globe Award (July 2019), recognised for its outstanding work and contribution towards advancing peer to peer learning in climate change adaptation among Pacific communities. The Energy Globe Award, also known as World Awards for Sustainability and Nature's Nobel Prize, is one of the most prestigious environmental awards worldwide.

Dr Mead's career is focussed on the application of environmental science for sustainable development and the management of environmentally beneficial projects, and, like his associates with eCoast, he actively seeks ways to provide clients and the community with well-balanced solutions to coastal management and development projects. He currently divides the majority of his professional time between Raglan NZ and Fiji.

SPECIALISED SKILLS/TRAINING AND OTHER BACKGROUND

- Company Director, Technical Group Manager, Project Manager, Project Leader/Field work programme leader, Designer – Managing Director/Director of ASR Ltd (1997-2011), Managing Director of eCoast (2011-present), Management of technical team and consulting/construction projects, Expert Witness, design and implementation of oceanographic data collection programmes for coastal process investigations and numerical model calibration, multi-purpose reef development/design (>40 projects), design of coastal structures and beach restoration/development projects, beach and marina design, beach management and resilience strategies for sea level rise, and ecology data collection programmes for ecological assessment and monitoring.
- Coastal Engineer/Scientist for the Tongan and Marshall Islands components of the Pacific Community's Global Climate Change Alliance: Pacific Small Island States (GCCA: PSIS), a programme which received the 2019 Energy Globe Award, recognised for its outstanding work and contribution towards advancing peer to peer learning in climate change adaptation among Pacific communities. The Energy Globe Award, also known as World Awards for Sustainability and Nature's Nobel Prize, is one of the most prestigious environmental awards worldwide.
- FETA Fijian Tourism Awards – Sustainable Tourism 2016, and Culinary Excellence 2018 (Maqai Beach Eco Resort (Eco Surf Viti Ltd))
- Entrepreneur of the Year Finalist, 2009
- While Managing Director of ASR Ltd:
 - 10th fastest growing company in New Zealand (Deloitte's Fast 50)
 - Fastest exporter company in central North Island (Deloitte's)
 - Fastest growing technical, media, telecommunications company in central North Island (Deloitte's)
 - ANZ Waikato Export Awards 2006 Innovator of the Year (ASR Ltd).
- Computer modelling (the 3DD suite) - WBEND (wave refraction and beach erosion/deposition model), GENIUS (sediment transport), 3DD (3-dimensional hydrodynamic and sediment transport), 2DBEACH (coupled beach and sediment transport model), Pol3DD ((POLlution dispersal coupled to 3DD – sediment transport, pollutants, larvae, etc.). Data analysis. Experience with a wide range environmental parameter testing procedures and equipment (water – nutrients, chemical properties, biological properties, etc., air, sediment, etc.)

- 4th Year Environmental Law (NZ RMA 1991)
- Experienced with a variety of software applications especially in areas useful for numerical modelling, GIS and statistical analysis of data including; Surfer, AutoCAD, Grapher, SigmaStat/Plot, Matlab, Statistica, Basic FORTRAN, Mocha, Kaliedograph, Pathfinder.
- Experienced with the deployment and data analysis of a variety of oceanographic data acquisition equipment - GPS (a wide range of systems), sidescan sonar, S4, Sontek ADP, FSI, Dobie, Nortek (Aquadopps and Profilers), Sentinel, CTD's, drones (overhead and underwater), etc.
- Lecturer – Coastal Engineering, Environmental Change, AutoCAD, Physics
- Rescue diver (>2500 field-work dives)
- Day Skipper's Certificate/Restricted Radiotelephone Operator's Licence
- Martial Arts Instructor (3rd Dan)
- Senior Science Award (University of Auckland)
- Paton Cup – Highest Marks in New Zealand School Certificate Technical Drawing

Peer-Reviewed Papers: 60

Conference Papers: 14

Technical Reports: >450

International Conference Presentations: 28

National Conference Presentations: 28

Seminars and Public Presentations: 56

Expert Witness/Evidence: 66

Popular Articles: 33

Graduate Student Supervision: 17

Keynote Presentations: 11

Book Chapters: 2

NATIONAL AND INTERNATIONAL CONFERENCES ATTENDED:

- Global Wave Conference 2020. Gold Coast, Queensland, Australia, 10-14 February 2020.
- New Zealand Associate of Impact Assessments (NZAIA) Conference. Auckland, New Zealand, 21-23 November 2019.
- New Zealand Coastal Society (ENZ) Conference. Invercargill, New Zealand, 13-15 November 2019.
- New Zealand-Fiji and Fiji-New Zealand Business Councils Annual Joint Conference, Auckland, New Zealand, 29 August 2019.
- Fiji Trade and Investment Roadshow – Presentations on Sustainable Tourism and Environmental Consulting in Fiji. Wellington and Auckland, 6 & 8 May 2019.
- New Zealand Coastal Society (ENZ) Conference. Gisborne, New Zealand, 21-23 November 2018.

- Fiji-New Zealand and New Zealand-Fiji Business Councils Annual Joint Conference, Suva, Fiji, 25-28 June 2018.
- New Zealand Coastal Society (IPENZ) Conference. Tauranga, New Zealand, 15-17 November 2017.
- The 23rd Australasian Coasts and Ports Conference, Cairns, Australia, 20-23 June 2017.
- Fiji-New Zealand and New Zealand-Fiji Business Councils Annual Joint Conference and 30th Anniversary, Auckland, New Zealand, 15 June 2017.
- New Zealand Coastal Society (IPENZ) Conference. Dunedin, New Zealand, November 2016.
- The 13th International Coral Reef Symposium, Honolulu, Hawaii, 20-25 June 2016.
- The 22nd Australasian Coasts and Ports Conference, Auckland, New Zealand, 15-18 September, 2015.
- Fiji-New Zealand and New Zealand-Fiji Business Councils Annual Joint Conference, Auckland, New Zealand, 12 June 2015.
- New Zealand Coastal Society (IPENZ) Conference. Raglan, New Zealand, November 2014
- Fiji-New Zealand and New Zealand-Fiji Business Councils Annual Joint Conference, Suva, Fiji, 28 June 2014.
- The 21st Australasian Coasts and Ports Conference, Manly, Australia, September, 2013
- New Zealand Climate Change Conference, Palmerston North, 4-5 June 2013
- New Zealand Coastal Society (IPENZ) Conference. Auckland, New Zealand, November 2012
- ASBPA 2012 National Coastal Conference. San Diego, October 2012
- New Zealand Fiji Business Council Conference, Nadi, 16-17 June, 2012
- Sea-Level Rise – Meeting the Challenge, Wellington 10-11 May 2012.
- New Zealand Coastal Society (IPENZ) Conference. Nelson, New Zealand, November 2011.
- The 20th Australasian Coasts and Ports Conference, Perth, Australia, September, 2011
- New Zealand Coastal Society (IPENZ) Conference. Whitianga, New Zealand, November 2010.
- 32nd International Conference on Coastal Engineering. Shanghai, China, June 30 – July 5, 2010.
- Australasian Young Planners Conference VPCoast2010. Christ Church, New Zealand, 19-20 April 2010. 7th International Multi-Purpose Artificial Surfing Reef Symposium, Bondi Beach, Sydney, Australia, March 19th, 2010.
- The 19th Australasian Coasts and Ports and NZ Coastal Society (IPENZ) Conference, Wellington, New Zealand, September 15-18th, 2009
- 6th International Multi-Purpose Artificial Surfing Reef Symposium, Jeffrey's Bay, South Africa, May 18-21st, 2009
- New Zealand Coastal Society (IPENZ) Conference. New Plymouth, New Zealand, November 2008.
- New Zealand Coastal Society (IPENZ) Conference. Tauranga, New Zealand, November 2007.
- The 18th Australasian Coasts and Ports Conference, Melbourne, Australia, July 2007
- Inaugural Conference of The Dune Restoration Trust of New Zealand. Tauranga, New Zealand, 13-17 February, 2007

- New Zealand Coastal Society (IPENZ) Conference. Kaikoura, New Zealand, 15-17 November 2006.
- 5th International Surfing Reef Symposium, Heaven on the Planet, Lombok, Indonesia, July 31-August 3, 2006
- The 3rd Western Australian Coastal Conference. Bunbury, West Australia, 16-18 November 2005
- New Zealand Coastal Society (IPENZ) Conference. Tutukaka, New Zealand, 12-14 October 2005.
- The 17th Australasian Coasts and Ports Conference, Hilton Hotel, Adelaide, Australia, 21-23 September 2005
- The 4th International Surfing Reef Conference. Manhattan Beach, California, 11-14 January 2005.
- New Zealand Coastal Society (IPENZ) Conference. Dunedin, New Zealand, 18-20 October 2004.
- The 2nd Western Australian Coastal Conference. Geraldton, West Australia, 20-21 November 2003.
- SASIC 3 - Third Surfing Arts, Science and Issues Conference. Doheny Doubletree Inn, Dana Point, California, USA, 8-9 November 2003
- The 16th Australasian Coasts and Ports Conference, Hyatt Regency Hotel, Auckland, New Zealand, 9-12 September 2003
- The 3rd International Surfing Reef Conference. Karioi Centre, Raglan, New Zealand, 23-25 June 2003.
- SASIC 2 - Second Surfing Arts, Science and Issues Conference. Holiday Inn, Ventura, California, USA, 9 November 2002
- New Zealand Marine Science Conference, Rutherford Hotel, Nelson, New Zealand, September 2-4 2002
- New Zealand Coastal Society (IPENZ) Annual Symposium, Tahuna Beach Conference Centre, Nelson, New Zealand, October 17-18 2001
- The 15th Australasian Coasts and Ports Conference, Gold Coast, Australia, Sept 28-1 Oct 2001
- New Zealand Marine Science Conference, Waikato University, Hamilton, August 2000
- International Coastal Symposium 2000, Sheraton Hotel, Rotorua, New Zealand, April 24-27 2000
- New Zealand Marine Science Conference, Victoria University, Wellington, August 1999
- The 14th Australasian Coasts and Ports Conference, Perth, Australia, April 1999
- Joint Australasian Botany Society/New Zealand Marine Sciences Society Conference, University of Otago, Dunedin, July, 1998
- 2nd Annual International Artificial Surfing Reef Symposium, San Diego, April 1998
- Joint New Zealand/Australia Marine Science Conference, Auckland, July 1997
- New Zealand Marine Science Conference, University of Canterbury, Christ Church, August 1996
- New Zealand Marine Science Conference, Victoria University, Wellington, August 1995

PUBLICATION SUMMARY

Peer-reviewed publications

Atkin, E., S. T. **Mead** and D. J. Phillips 2019. Investigations of Offshore Wave Preconditioning. Submitted to the Journal of Coastal Research (Special Issue 87: Surf Break Management in Aotearoa New Zealand).

Mead, S. T., and T. Haggitt, 2019. Coastal Engineering Construction Impact Monitoring of Rangitahi Bridge, New Zealand, and Climate Change Resilience in Eastern Tongatapu. Proceedings of the Australasian Coasts & Ports 2019 Conference – Hobart, Australia 10-13 September 2019.

Orchard, S., Atkin, E.A. and **Mead** S.T., 2018. Development of the Regional Significance Concept for Surf Break Management in Aotearoa New Zealand. Submitted to the Journal of Coastal Research (Special Issue 87: Surf Break Management in Aotearoa New Zealand)

Atkin, E.A, Bryan, K., Hume, T., **Mead**, S.T., and Waiti, J., 2018. Management Guidelines for Surfing Resources. Raglan, New Zealand: Technical Group on Surfing Resources.

Hume, T.M., N. Mulcahy and S. T. **Mead**, 2018. An overview of the breaking wave environment in New Zealand – Use and values. Submitted to the *Journal of Coastal Research* (Special Issue 87: *Surf Break Management in Aotearoa New Zealand*)

Mead, S. T., and E. Atkin, 2018. Managing Issues at Aotearoa New Zealand's Surf Breaks. Submitted to the *Journal of Coastal Research* (Special Issue 87: *Surf Break Management in Aotearoa New Zealand*)

Phillips, D. J., S. T. **Mead**, and M. Emeny, 2017. Lyall Bay Coastal Remediation. *Proceedings of the 23th Australasian Coasts and Ports Conference*, Cairns, Australia, 21-23 June 2017.

Atkin, E., T. Hume, S. **Mead**, K. Bryan, and J. Waiti, 2017. Remote Sensing, Classification and Management Guidelines for Surf Breaks of National and Regional Significance. *Proceedings of the 23th Australasian Coasts and Ports Conference*, Cairns, Australia, 21-23 June 2017.

Mead, S. T., D. J. Phillips, and E. Atkin, 2017. The Rise and Fall (and Rise) of Winston Island. *Proceedings of the 23th Australasian Coasts and Ports Conference*, Cairns, Australia, 21-23 June 2017.

Borrero J. C., M. Clarke, R. Klaus, S. T. **Mead** and S. Persand. Design and Assessment of Climate Change Adaptation and Erosion Control Measures for Mon Choisy Beach, Republic of Mauritius. *Proceedings 13th International Coral Reef Symposium*, Honolulu, Hawaii, 20-25 June 2016.

Haggitt, T., and S. T. **Mead**, 2015. Makara Estuary Monitoring: Effects-based monitoring within a degraded, yet dynamic, coastal environment. *Proceedings of the 22th Australasian Coasts and Ports Conference*, Auckland, New Zealand, 15-18 September 2015.

Mead, S. T., J. C. Borrero, E. Atkin and D. J. Phillips, 2015. Application of Climate Change Adaptation, Resilience, and Beach Management Strategies on Coral Islands. *Proceedings of the 22th Australasian Coasts and Ports Conference*, Auckland, New Zealand, 15-18 September 2015.

Mead, S. T., D. Phillips and A. Prime, 2013. Development of a Multi-Purpose Breakwater/Reef at Maqai Eco Surf Resort, Qamea Island, Fiji. *Proceedings of the 21th Australasian Coasts and Ports Conference*, Sydney, Australia, 11-13 September 2013.

Borrero, J. C., J. Oldman, L. Lebreton, S. T. **Mead**, and D. James, 2012. Assessment of Submerged Structures for Coastal Protection in a Low Wave Energy Environment. In: Innovative Coastal Zone Management: Sustainable Engineering for a Dynamic Coast, January 2012.

Mead, S. T., D. J. Phillips, and T. Haggitt, 2011. Development of a GIS to Determine the Vulnerability of Regionally Significant Marine Receiving Environments to Land-Use Impacts. *Proceedings of the 20th Australasian Coasts and Ports Conference*, Perth, Australia, 27-30 September 2011.

- Atkin, E. A., J. C. Borrero and S. T. **Mead**, 2013. Morphological Response to a Multi-Purpose Reef. *Journal of Coastal Engineering*, (Submitted).
- Mead**, S. T., and J. C. Borrero, 2011. Multi-Purpose Reefs – A Decade of Applications. *Proceedings of the 20th Australasian Coasts and Ports Conference*, Perth, Australia, 27-30 September 2011.
- Borrero, J. C., S. T. **Mead** and A. Moores, 2010. Stability Considerations and Case Studies of Submerged Structures Constructed from Large, Sand-Filled, Geotextile Containers. *Proceedings of 32nd International Conference on Coastal Engineering*. Shanghai, China, June 30 – July 5, 2010.
- Mead** S.T, C. Blenkinsopp, J. C. Borrero and A. Moores, 2010. Design and Construction of the Boscombe Multi-Purpose Reef. *Proceedings of 32nd International Conference on Coastal Engineering*. Shanghai, China, June 30 – July 5, 2010.
- Scarfe, B., T. Healy, H. Rennie and S. **Mead**, 2009. Sustainable Management of Surfing Breaks: Case Studies and Recommendations. *Journal of Coastal Research*, 25(3): 684-703.
- Harrison, S., J. Borrero, C. Klinginger, S. T. **Mead** and D. Phillips, 2009. Hydrodynamic Modelling of Whaingaroa Harbour. *Proceedings of the 19th Australasian Coasts and Ports Conference, Wellington, NZ*, 16-18 September 2009.
- Phillips D., S. T. **Mead**, S. Harrison J. Frazerhurst, G. Dodet, C. Klinginger and J. Borrero, 2009. Oceanography in the Public Interest: Tales from Raglan. *Proceedings of the 19th Australasian Coasts and Ports Conference, Wellington, NZ*, 16-18 September 2009.
- Black, K.P., A.E. Moores, and S.M. **Mead**, 2009. Effect of Input Wave Shape on Surfing Wave Quality. *Coastlab08 - Application of Physical Modelling to Port and Coastal Protection*. Publisher: IAHR. ISBN: 78-90-78046-07-3.
- Mead**, S. T., 2009. *Multiple-Use Options for Coastal Structures: Unifying Amenity, Coastal Protection and Marine Ecology*. The Reef Journal, Vol. 1, 2009. ISSN No. 1176-7812.
- Phillips, D. J., and S. T. **Mead**, 2008. Investigation of a Large Offshore Sandbar at Raglan, New Zealand: Impacts on Surfing Amenity. *Shore and Beach* Vol 76(2) Spring 2008
- Black K. P., and S. T. **Mead**, 2007. Sand bank responses to a multi-purpose reef on an exposed sandy coast. *Shore and Beach* 75(4):55-66.
- Mead**, S. T., J. C. Borrero, K. P. Black and D. Anderson, 2007. Multi-Faceted Beach Management at St Francis Bay Beach, South Africa. *Shore and Beach* 75(4):43-54.
- Mead** S. T., and D. J. Phillips, 2007. Temporal and Spatial Variation of a Large Offshore Sandbar at Raglan, NZ. *Proceedings of the 18th Australasian Coasts and Ports Conference*, Melbourne 2007.
- Mead** S. T., K. P. Black and A. Moores, 2007. Mount Maunganui Reef – Amalgamating Design and the Constraints of Construction. *Proceedings of the 17th Australasian Coasts and Ports Conference*, Melbourne 2007.
- Longdill, P. C., T. R. Healy, K. P. Black and S. T. **Mead**, 2007. Integrated Sediment Habitat Mapping for Aquaculture Zoning. *Journal of Coastal Research*, Special Issue 50, *Proceedings of the International Coastal Symposium*, Australia.
- Mead**, S. T., and K. Black, 2006. Innovative Shoreline Protection for Oil Piers, Ventura, (USACE Section 227 Demonstration Program). 30th International Conference on Coastal Engineering, September 3-8, 2006. Manchester Grand Hyatt San Diego, San Diego, California.

- Phillips, D. J., and S. T. **Mead**, 2006. *Investigation of a Large Sandbar at Raglan, New Zealand: Project Overview and Preliminary Results*. 5th International Surfing Reef Symposium, Lombok, Indonesia, July 31-August 3, 2006. Volume 1: Reef Journal ISSN No:1176-7812.
- Moore, A., K. P. Black and S. T. **Mead**, 2006. *Physical Modelling of the Mount Maunganui Artificial Surfing Reef*. CoastLab06, Porto, Portugal.
- Mead**, S. T., and K. P. Black, 2005. Development of a Multi-Purpose Reef at Orewa Beach, New Zealand. *Proceedings of the 17th Australasian Coasts and Ports Conference*, Adelaide.
- Scarfe, B., Rennie, H, S. T. **Mead**, T. R. Healy and C. Nelson, 2005. Sustainable Management of Surfing Breaks – An Overview. 4th International Artificial Surfing Reef Symposium. Volume 1: Reef Journal ISSN No:1176-7812.
- Black, K. P., and S. T. **Mead**, 2005. Design of Surfing Reefs. 4th International Artificial Surfing Reef Symposium – Volume 1: Reef Journal ISSN No:1176-7812.
- Mead**, S. T., K. P. Black, B. Scarfe, L. Harris and C. Blenkinsopp, 2005. Detailed Design of a Multi-Purpose Reef at Oil Piers, Ventura, California. 4th International Artificial Surfing Reef Symposium – a Special Issue of the *Journal of Coastal Research*. Submitted.
- Erftemeijer, P. L. A., R. F. de Graaff, S. T. **Mead** and G. Boot, 2004. Site-Selection for Artificial Reefs in Bahrain Based on GIS-Technology and Hydrodynamic Modelling. *Ocean & Coastal Management*.
- Black and **Mead**, 2003. Numerical Prediction of Salient Formation in the Lee of Offshore Reefs. *Proceedings of the 3rd International Surfing Reef Symposium, Raglan, New Zealand, June 22-25, 2003*. Pp 196-218.
- Mead**, S. T., and P. McComb, 2003. Remote Video Sensing to Support Ecological Impact Assessment: the correlation of habitat complexity and species diversity/abundance allows for confident assessment of large subtidal areas. *Proceedings of The 16th Australasian Coasts and Ports Conference*, Hyatt Regency Hotel, Auckland, New Zealand, 9-12 September 2003
- Frazerhurst, J., and S. T. **Mead**, 2003. Determination of Optimum Wave Energy Converter Device Locations Utilising Numerical Modelling of Wave Transformation. *Proceedings of the Fifth European Wave Energy Conference*, Cork, Ireland, 17-20 September, 2003.
- McComb, P., K. P. Black and S. T. **Mead**, 2003. *A Surfing Reef Feasibility Study at Opunake, New Zealand*. 3rd International Artificial Surfing Reef Symposium, Raglan, New Zealand 23-25 June 2003. ISBN 0-473-09801-6 2003
- Phillips, D. J., T. R. Healy, K. P. Black and S. T. **Mead**, 2003. *Surf Zone Currents and Influence on Surfability*. 3rd International Artificial Surfing Reef Symposium, Raglan, New Zealand 23-25 June 2003. ISBN 0-473-09801-6 2003
- Hadersdorfer, H., K. P. Black and S. T. **Mead**, 2003. *Floating Reefs for Surfing Pools*. 3rd International Artificial Surfing Reef Symposium, Raglan, New Zealand 23-25 June 2003. ISBN 0-473-09801-6 2003
- Scarfe, B., M. H. S. Elwany, S. T. **Mead** and K. P. Black, 2003. *Categorising the Types of Surfing Breaks Around Jetty Structures*. 3rd International Artificial Surfing Reef Symposium, Raglan, New Zealand 23-25 June 2003. ISBN 0-473-09801-6 2003
- Scarfe, B., M. H. S. Elwany, S. T. **Mead** and K. P. Black, 2003. *The Science of Surfing Waves and Surfing Breaks: A Review*. 3rd International Artificial Surfing Reef Symposium, Raglan, New Zealand 23-25 June 2003. ISBN 0-473- 09801-6 2003

Burgess, S. C., K. P. Black, S. T. **Mead**, M. J. Kingsford, 2003. *Considerations for Artificial Surfing Reefs as Habitat for Marine Organisms*. 3rd International Artificial Surfing Reef Symposium, Raglan, New Zealand 23-25 June 2003. ISBN 0-473-09801-6 2003

Mead, S. T., K. P. Black, J. Frazerhurst and B. Scarfe, 2003. *The Effects of Wave Focussing on Surfing Reef Site Selection, Surfing Wave Quality and ASR Design at Scales of Inner Continental Shelf to Sub-Tidal Reef*. 3rd International Artificial Surfing Reef Symposium, Raglan, New Zealand 23-25 June 2003. ISBN 0-473-09801-6 2003

Mead, S. T., 2003. *Surfing Science*. Proceedings of The 3rd International Surfing Reef Conference. Karioi Centre, Raglan, New Zealand, 23-25 June 2003. ISBN 0-473-09801-6 2003

Scarfe, B. E., A. K. Chong, W. P. de Lange, S. T. **Mead** and K. P. Black, 2003. Metric and Non-Metric Application of a Two-Dimensional Projective Coordinate Transformation for Coastal Zone Studies. *Photogrammetric Engineering and Remote Sensing*.

Scarfe, B. E., K. P. Black, A. K. Chong, W. L., de Lange, D. Phillips and S. T. **Mead**, 2003. The Application of Surveying Techniques to Artificial Surfing Reef Studies. *Trans-Tasman Surveyor*, April Edition, 2003.

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