FOT
Carrying capacity

Environment
Waikato
REGIONAL COUNCIL

Auckland
Regional Council
TE RAUHITANGA TAIAO

Factors Related to the Sustainability of Shellfish Aquaculture Operations in the Firth of Thames: a Preliminary Analysis

N. Broekhuizen

J. Zeldis

S. A. Stephens

J.W. Oldman

A.H. Ross

J. Ren

M.R. James

Prepared for

Environment Waikato & Auckland Regional Council

Contents

Reviewed by: Approved for release by:

JM Bradford-Grieve   M R James

Formatting Checked

....

Executive Summary

The Waihou and Piako Rivers, which drain pastoral catchments of Waikato and flow into the Firth of Thames, have important effects on the fluxes of water, salt and nutrients through the Hauraki Gulf. Current work is aimed at improving estimates of these mixing and biogeochemical processes.

The Hauraki Gulf and especially the Firth are the main spawning grounds for New Zealand's largest snapper fishery. John dory, rig, school shark, and barracouta juveniles are also relatively abundant within the Firth.

Average phytoplankton concentrations within the Firth of Thames are higher than those within the Marlborough Sounds. Thus, mussel growth rates in the Firth of Thames may be higher than those in the Marlborough Sounds.

Phytoplankton community structure, biomass and production show substantial variability – related both to ENSO effects on upwelled nutrient supply and to variation in water column structure within the Firth itself. This variability may influence the performance of shellfish farming operations.

The survival of larval fish may suffer if shellfish farming operations lead to wide-spread reductions in the abundance of zooplankton in the Firth, or if the shellfish themselves remove fish eggs and larvae from the water-column through filtering activities. Such effects are difficult to quantify at present.

The Firth of Thames and Hauraki Gulf is a net consumer of organic material (animals consume more organic material than phytoplankton produce). Furthermore, average phytoplankton concentrations are probably insufficient to support maximal growth rates amongst the herbivorous filter-feeding community. These two observations may imply that, if sufficiently wide-spread, even small reductions in phytoplankton abundance are likely to increase any food-limitation which other trophic groups suffer.

We offer a list of candidate characteristics by which 'ecological sustainability' may be measured (Table 5). We also offer suggestions as to the means by which the ecological significance of deviations from the local norm for each characteristic could be determined.

Tracer studies with biologically inert particles indicate that materials disperse away from farms more rapidly in the deeper waters of the northern Firth than they do in the southern Firth. The majority of dispersal is parallel with the Firth's estuary-ocean axis. Nonetheless, regardless of the point of origin, biologically inert particles will become mixed throughout the Firth within days to weeks under typical wind conditions.

Comparison of this mixing period with typical turnover times for a variety of materials which shellfish might influence (nutrients, phytoplankton, protozoa, mesozooplankton, fish eggs and larvae) suggests that, even in the absence of mixing with water which has not passed through a farm, the footprint of an **individual** farm may be relatively small (100s – 1000s of m radius) for highly reactive materials (nutrients, phytoplankton, protozoa), but will extend over the entire Firth for less reactive materials. Provided that the abundance of farms is not too high at the meso-scale (a spatial scale of a few-times the farm size), mixing between waters which have- and have not passed through the farm will reduce the area over which a farm's influence is detectable.

Depletion calculations suggest that, in the northern Firth of Thames a 50 ha farm may consume >20% of the phytoplankton which pass through it. If this loss is applied over the volume of the farm (rather than the volume of water passing through the farm), it equates to a depletion of approximately 1 – 2% of the total phytoplankton.

Improved estimates of the Firth's production capacity could be arrived at by applying both the ECOSIM/ECOPATH approach and adapting NIWA's model of mussel production capacity in the Marlborough Sounds.

Improved estimates of the Firth's ecological capacity require: (a) experimental effort related to mussel filtering (specifically, better characterisation of which zooplankton and mero-plankton species mussels are able to filter from the water-column), (b) application of existing simulation models over a wider range of environmental conditions, (c) enhancements to existing simulation models.

1 Introduction

1.1 Aims and Structure of this report

Over the past decade shellfish farming operations in New Zealand have grown rapidly. Most of this growth has been in the cultivation of the Greenshell™ mussel, *Perna canaliculus*, though significant numbers of Japanese Oyster *Crassostrea gigas* are also farmed. In the year 2000, approximately 2500 ha of New Zealand's coastal seabed were allocated to mussel production (Inglis et al. 2000), yet there are pending applications for approximately 6000 ha of mussel farms within the Firth of Thames alone! Furthermore, many of the pending applications for the Firth of Thames (and elsewhere in New Zealand) are for farms which are at least an order of magnitude larger than any farm presently operating in New Zealand.

Development of marine aquaculture operations has been regulated by regional authorities through application of the Resource Management Act; however, to date this has tended to be an *ad-hoc* process with few formalised guiding principals. Central Government has recently proposed a two year moratorium upon further aquaculture development in order to provide an opportunity in which: (a) government and regional authorities can commission the research which they deem necessary to improve the scientific foundations upon which judgements of sustainability are to be made, and (b) develop appropriate zoning policies and aquaculture management areas (AMAs) for their coastal waters.

This report (commissioned by Environment Waikato and the Auckland Regional Council) complements Hatton et al. (2002) (commissioned by Environment Waikato). Hatton et al. (2002) (a) presents an overview of existing farms within the Firth of Thames, (b) reviews the domestic and international approaches to monitoring and performance standards for shellfish aquaculture, (c) assesses how these might be applied to the Firth of Thames, (d) suggests a process by which trigger-levels and environmental standards for aquaculture might be developed, (e) identifies some characteristics which might be used to assess performance.

Within this report our aims are to:

1. review relevant data on the Firth of Thames.

2. Highlight issues which are specific to the Firth of Thames.

3. Discuss approaches to the assessment of the sustainability of aquaculture – with emphasis on the types of data and models which are required.

4. Provide an initial assessment of sustainability of aquaculture within the Firth of Thames based upon data which are currently available.

5. Provide examples of what further improvements upon the initial assessment might be gained with additional funding (using presently available tools).

6. Summarise the data and modelling requirements for an improved estimate of the impacts of aquaculture in the Firth of Thames.

7. Develop a research plan to yield the required additional data and enhanced numerical models which will make it possible to derive more robust estimates of the sustainability of aquaculture activities within the Firth of Thames.

Within this report, sections 2 – 6 correspond to the first four of these themes.   The remaining three themes are addressed in section 7.

Within the remainder of this introductory section, we introduce some terminology and briefly review the effects which may be associated with shellfish aquaculture.

1.2   Terminology

This report concerns the sustainability of shellfish aquaculture within the Firth of Thames. We choose to define the northern limit of the Firth of Thames by a line drawn between Coromandel township and Waiheke Island, then south to the North Island mainland circa Kawakawa bay (see Bradley & Terralink NZ Ltd 1999, and Figure 3). Nonetheless, there are substantial exchanges of water and other materials between the Firth of Thames and the Hauraki Gulf – onto which the Firth opens. Thus, we will also make extensive reference to data from the Hauraki Gulf within this report. We will use the term **Greater Hauraki Gulf** to refer to the Firth of Thames and Hauraki Gulf collectively.

Inglis *et al*. (2000) define four types of carrying capacity relevant to the management of coastal aquaculture:

a. **Physical carrying capacity** relates to restrictions placed upon the size and number of farms by factors such as geography, planning restrictions, infrastructure

b. **Social carrying capacity** relates to limits arising from impacts which are deemed to be socially unacceptable. These might include reduced visual amenity, and access restrictions.

c. **Production carrying capacity** is defined as the stocking density which allows the sustainable harvest to be maximised.

d. **Ecological carrying capacity** is defined as the quantity of farm development beyond which ecological impacts are deemed to become unacceptable.

The former two types of carrying capacity have substantial components which lie outside NIWA's area of expertise. Thus, in the remainder of this report we make only brief reference to them and focus upon the latter two types of carrying capacity.

Shellfish production is relatively easily measured. Consequently, the concept of production carrying capacity is readily appreciated. Nonetheless, it is important to realise that, given the common ownership rights (*cf* individual ownership rights) which the farm operators hold in the resource which their shellfish will exploit (i.e. phytoplankton), there is ample scope for conflict between individual operators, each seeking to maximise their own yield with little regard to any detrimental effects upon other operators. This conflict is not unique to mussel farming and the processes that lead to it have been termed **"The Tragedy of the Commons"** (Hardin 1968). An important consequence of the **Tragedy of the Commons** is that the behaviour which allows an individual to maximise his or her own return is rarely the behaviour which will allow the community as a whole to maximise the sum of individual returns. Thus, when discussing production carrying capacity, it is important to define whether one is referring to this with respect to the comparatively small scale of an individual farm / operator, or with respect to a regional scale (in which there may be several farms / operators).

Like production carrying capacity, ecological carrying capacity is conceptually easy to appreciate. A corollary of the definition of ecological carrying capacity is that farms which do not exceed this capacity are deemed to be ecologically sustainable. Those which do exceed the ecological carrying capacity are deemed ecologically unsustainable. Unfortunately, unlike production carrying capacity, there are no universally agreed characteristics by which to determine how far a system is from its ecological carrying capacity (or even, what this capacity is). Even where such characteristics can be agreed upon, there are no agreed methods by which to determine what degree of deviation from the ecological carrying capacity is unacceptable. If aquaculture development is to be managed with respect to ecological carrying capacity issues, it will be necessary to convene a group with responsibility for defining appropriate characteristics and corresponding threshold levels of change at which further investigations, or intervention would become triggered (Hatton et al. 2002). We make some further recommendations on this issue within Section 4 of this report.

Throughout this report, we make extensive use of phrases such as 'effects of a farm' and 'impacts of aquaculture'. This use is a convenient shorthand by which we are able to avoid repeated use of cumbersome phrases such as 'a measurable change in environmental conditions as a result of aquaculture activity'. We will use the terms **impacted** or **affected** to refer to waters (or characteristics of these waters, or the sea-floor below) that are measurably changed as a result of passage through a farm, or being under a farm. By analogy, we adopt the term **pristine** to refer to waters (or characteristics thereof, or the sea-floor below) which are unaffected by farming activities (because: i. they have not come into close proximity with a farm; ii. the farm simply did not modify this particular characteristic; iii. though previously modified by a farm, this characteristic has since recovered). We emphasize that we are adopting these all of these shorthand phrases as a convenience only. We use the terms impact and effect merely to indicate that a change can be detected. This need not imply that the change is (un)desirable or that it will have (un)desirable consequences. An assessment of the environmental value of any effect of impact depends not solely upon the absolute size (local intensity and spatial/temporal extent) of this impact, but also upon the environment's ability to 'absorb' an effect of this size. Thus, the environmental value of any effect requires not only an absolute measure of the size of the effect, but additional contextual data by which to assess the impact's relative size. Furthermore, we wish to emphasize that there is a distinction between a 'statistically significant' effect and an 'ecologically significant' one. The former merely indicates that the chance that the observed deviation between control and treatment could have arisen by chance alone is no greater than some specified level (traditionally, but arbitrarily, 5%). An effect is judged to be ecologically significant if is deemed to be large enough in magnitude, or spatial extent that it is, of itself, an indication of abnormal ecosystem function, or likely to lead to such abnormalities. Unfortunately, it is often difficult to derive an objective definition of what is abnormal.

1.3   Review of Documented Aquaculture impacts

In this section, we present a brief description of the impacts which can be associated with shellfish aquaculture in coastal waters. Readers who are seeking more complete descriptions are referred to Morrisey et al. (1996) and Inglis et al. (2000).

Impacts can be subdivided into two classes: local (ie within, below, and within a few tens of meters around a farm), and far-field (ie at distances beyond the local scale). They can also be classified into those that have been postulated to occur and, a more restricted subset that have been demonstrated to occur. Local effects have been extensively studied, and many of the postulated local impacts have also been demonstrated in practise. In contrast, the majority of far-field impacts remain no more than postulates at present. In part, this absence of confirmed far-field impacts reflects

the subtle nature of many of the proposed effects but more importantly, we are aware of no published studies which have been designed with a view to detecting far-field impacts. Nonetheless, the majority of far-field effects are merely 'wake-effects' – i.e., they reflect current-driven advective transport of materials (or deficits thereof) away from the farms. As such, the key issue is not **whether** these far field effects occur, but rather **how quickly** (in time and space) they are dissipated.

As water and associated materials are transported away from an individual farm, the impacts of that farm are dissipated. Two distinct processes drive dissipation: mixing and *in-situ* regeneration. Mixing between impacted and neighbouring, pristine water **dilutes** the impact (reduces its magnitude at the expense of further increasing its spatial extent). In contrast regenerative processes (such as phytoplankton growth) lead to recovery of the impacted water without entailing any increase in the spatial extent of the impact. The relative importance of the two processes is determined by their respective rates. These are discussed in more detail within Section 5, but it is worth emphasizing that the relative importance of mixing will decline as the surface area: volume ratio of the impacted water falls (ie, as the volume of an individual parcel of impacted water falls). This implies that the contribution (relative to regeneration) of mixing to the dissipation process is likely to be smaller around a large aquaculture facility than it is around a small one. Furthermore, it implies that mixing will become less important as the mesoscale (a few multiples of the farm-size) intensity of aquaculture operations rises (because there is a higher likelihood that a farm's impacted water will mix with that of another farm rather than with pristine water). Clearly, far-field impacts will be dependent upon not only upon the characteristics of each individual farm, but also upon the regional/meso-scale density and spatial distribution of farms.

In comparison with fin-fish aquaculture, shellfish farming is regarded as being relatively benign. This is largely because it has not relied upon either: supplementary feeding (with consequent deposition of unused food), or extensive use of pharmaceutical compounds to control diseases. Nonetheless, at the local-scale, there are well-documented effects associated with shellfish aquaculture (Table 1). Documented impacts include: localised phytoplankton and zooplankton depletion, changed sediment characteristics and changed nutrient dynamics. Aquaculture structures also provide a novel habitat. This is used by a wide variety of opportunistic species – including exotics and other undesirables. For example, the first records of the occurrence of the exotic sea-weed *Undaria pinnatifida* (Asian kelp) in the Marlborough Sounds were on mussel farm structures. Note, there have been no records of *U. pinnatifia* in the Firth of Thames region, but mussel farming may pose no risk in this respect because water-temperatures in the Firth of Thames and Hauraki Gulf are probably too high to enable substantial populations of this seaweed to become established (Sinner et al. 2000).

The rapid expansion of aquaculture suggests that, to date, the impacts of shellfish aquaculture have been deemed to be acceptable within New Zealand (and in many other

countries). It is, however, unclear to what extent this reflects a view that, even locally, the effects are not sufficiently severe to warrant concern, and to what extent it reflects the view that, though they may be locally severe, the effects are not sufficiently extensive in space to warrant concern. Nonetheless, given the number of pending applications (within the Firth of Thames, and elsewhere in New Zealand waters), it is clear that the total area of coastal waters suffering localised impacts will increase. Furthermore, given that many of the proposed farms are many times larger than existing farms, it plausible that local impacts will become more severe and far-field impacts more likely.

Table 1
Postulated and Documented local-scale effects associated with
shellfish aquaculture

Nature of Direct Impact	Possible consequences	References
Nutrient enhancement through shellfish excretion	Enhanced algal growth rates	(Gibbs et al. 1992)
Nutrient release from degrading faeces & pseudofaeces	Buffering of pelagic nutrient depletion Enhanced algal growth rates	(Tenore et al. 1982)
Oxygen depletion within water-column, or sediments	Physiological stress amongst planktonic organisms, emigration of larger (motile) organisms Pulsed release of nutrients and sulfides etc. from sediments	Considered unlikely in shellfish aquaculture operations (Morrisey & Swales 1996), but see
Change in particle size-spectra and particulate content in water-column	Changed sedimentation & sediment characteristics Changed light scattering	Impacts upon light field considered unlikely (Ross, A.H. 2002)
Removal of phytoplankton	Reduced food supplies for other phyto-herbivores Community composition biased towards fast-growing species	(Gibbs et al. 1992; Ogilvie et al. 2000) (Dahlbäck & Gunnarsson 1981)
Release of larval shellfish into water-column	Enhanced food-supply for some planktivores Other plankton may suffer greater competition for resources	Postulated
Depletion of zooplankton and eggs/larvae of fish & benthic invertebrates etc.	Other organisms may suffer greater competition for resources	(Tenore et al. 1985) (Horsted et al. 1988)
Accumulation of organic detritus and shell hash on sea-floor	Oxygen depletion Nutrient release, Changed in benthic species assemblage	(Dahlbäck & Gunnarsson 1981) (Grenz et al. 1990) (Kaspar et al. 1985)
Complexity contributed by the physical structure of the aquaculture facility and its biota	Changed species assemblage within the water-column and on the sea-floor (invasive species) Changed hydrodynamics Changed erosion/sedimentation characteristics	New Zealand experience with the ascidian Ciona intestinalis, seaweed Undaria pinnatfida and other mussels Mytilus galloprovincialis

Table 2

Postulated and documented far-field impacts of shellfish aquaculture

Nature of Impact	Possible Consequences	References
Changed current speeds	Enhanced local deposition leading to sediment starvation elsewhere	Postulated
Nutrient release by mussels and from organic debris	Enhanced phytoplankton growth, possibly with blooms	(Cembella et al. 1997)
Phytoplankton depletion	Selection for fast-growing species even where total far-field biomass is little changed	(Riemann et al. 1988) for changes in size structure within enclosures
	Adverse impacts upon phyto-planktivores	
Zooplankton depletion	Reduced zooplankton biomass, selection for fast growing species, adverse impacts upon higher trophic levels	(Horsted et al. 1988) for documented depletion of microzooplankton by *Mytilus edulis*. Higher order effects are speculative, but see this report for correlations between fish recruitrment and zooplankton abundance within the Firth of Thames
Depletion of eggs and larvae of fish and benthic invertebrates	Reduced species abundance, changed community structure and function	Postulated but see (Cummings et al. 2001)
Selection for species lacking an egg/larval dispersive stage		
Displacement of rocky shoreline/reef invertebrates by mussels through pre-emptive settlement and consumption of the dispersive egg/larval stages	Reduces species abundance and community diversity	Postulated

2 Review of Present Data

In this section (Section 2) we summarise those characteristics of the Firth of Thames that are relevant to the sustainability of shellfish aquaculture. Many of the issues that we will cover are generic (in the sense that they will be issues which would be relevant in any geographic location). In Section 3 of this report we will emphasize those issues that to be of special concern within the Firth of Thames.

This sections draws extensively (but not exclusively) upon both published and unpublished data from the archives of NIWA and its predecessors. In particular, much of the data stems from the FRST funded programs **Biological effects of cross-shelf water transfer and Biophysical Interactions across the continental shelf**.

Phytoplankton are central to any discussion of the sustainability of shellfish aquaculture. They are the primary prey not only of shellfish, but also, directly, or indirectly of most of the other animals living in marine environments. If shellfish aquaculture were to induce changes in the abundance or composition of the phytoplankton community, it is likely that other changes to the ecosystem would follow. We therefore devote a large section of this review to a discussion of the factors which determine phytoplankton abundance and productivity in the Firth of Thames. In addition, we place substantial emphasis upon zooplankton – which are responsible for the transfer of phytoplankton production up into the higher trophic levels (such as fish and sea-birds) which are of more immediate interest to the general public and other parties. The Firth of Thames is the basis of important commercial recreational fisheries and cultural fisheries, and we discuss evidence suggesting that fish-recruitment within the Firth of Thames depends upon the abundance and composition of the zooplankton community. Before entering into these discussions, we will however start by summarising the physical conditions of the Firth of Thames and reporting upon the extent to which it is coupled with (subsidised by-, or subsidising neighbouring areas – notably the Hauraki Gulf).

2.1 Physical Characteristics

The Firth of Thames is a large, shallow estuary (Figure 1). At its southern end, it is fed by several rivers whilst it opens onto the Hauraki Gulf at its northern end. Water depths vary between 0 and 10 m in the southern half of the Firth but increase to between 10 and 40 m further to the north (Figure 1). Muds dominate the floor of most of the estuary, but these give way to sands as it opens onto the Hauraki Gulf (Figure 2).

Figure 1.

Bathymetry map of the Firth of Thames resolved to the scale used in our numerical model of circulation patterns (see Section 5). Also shown are the tracer release locations 1 to 6 (see Section 5).
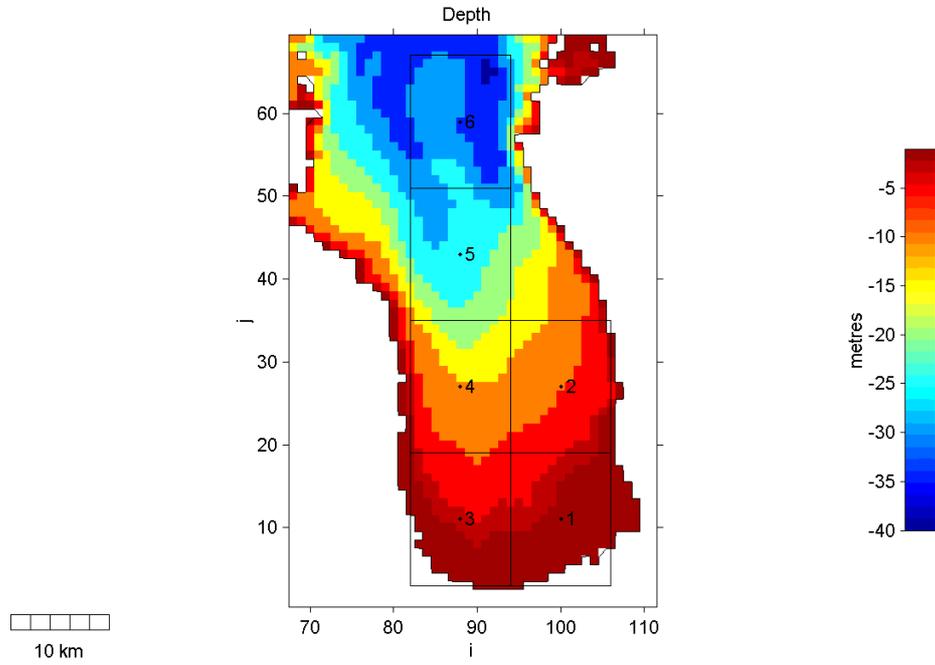
Figure 2.

Sea-floor sediment characteristics (Carter & Eade (1980), cited in kendrick & Francis (in review))
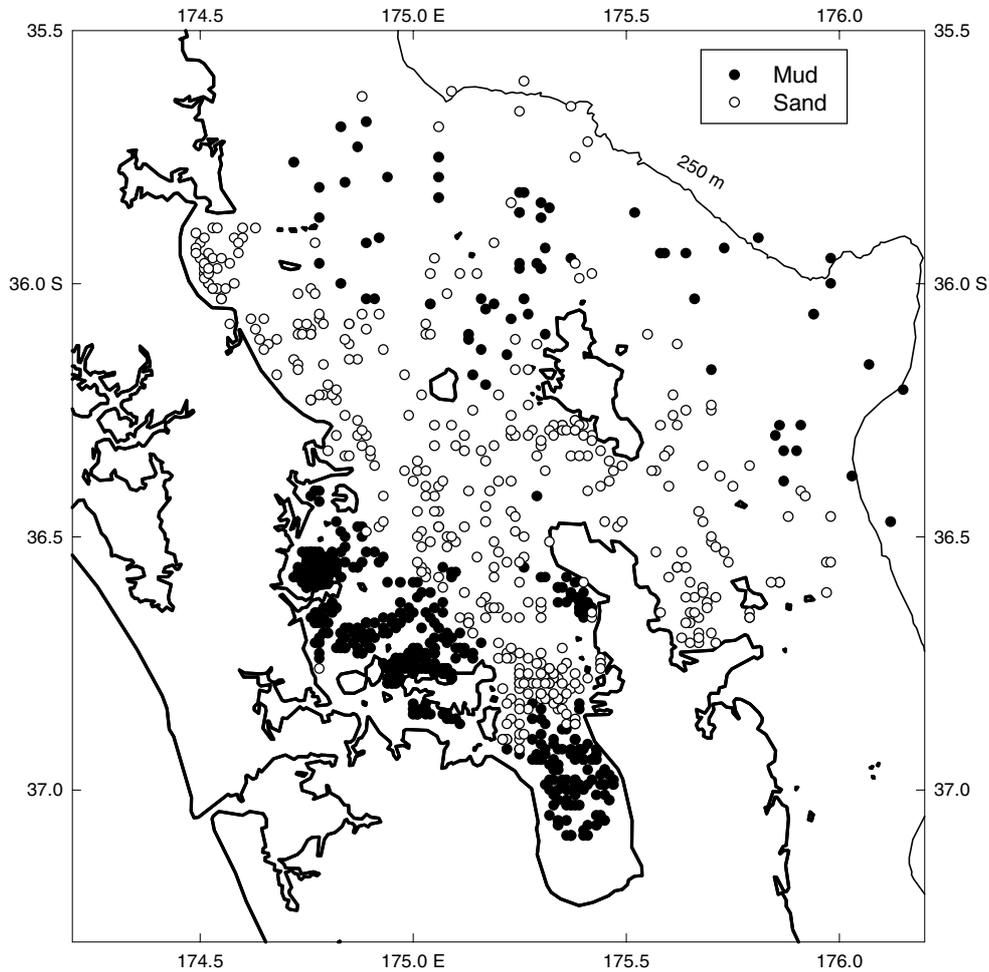
2.2   Mixing and fluxes of water and nutrients

We have already alluded to the role which mixing and transport are likely to play in distributing and dissipating any farm impacts (Section 1.3).   Mixing and transport also have direct impacts upon the abundance and productivity of phytoplankton and zooplankton. For example, currents determine the degree to which the Firth of Thames can be considered in isolation from the Hauraki Gulf. Having determined the extent of coupling between the Firth and the Gulf, we can also determine whether the Firth is a net exporter of material to the Gulf, or a net importer of material from the Gulf. If it proves to be a net importer it is less likely that changes occurring within the Firth of Thames will have consequences in the Gulf.

Numerical models of water circulation within the Firth of Thames and Hauraki Gulf (Black et al. 2000; Oldman & Senior 2000) demonstrate that although tides are strong in the Firth, the component of the net residual current driven by tides is weak (see also, sections 5 & 6 of this report). This is because there is little asymmetry between the ebb and flood tidal stages. It is winds that dominate the residual currents. In other words, if wind conditions are calm, there is little net movement of water from one tidal cycle to the next, whereas if winds are moderate or strong, there is considerable net movement related to wind direction, over the tidal cycle (see Section 5 for more details). The tides, however, are important because they provide energy to stir the water and disperse suspended and dissolved materials. The degree of stratification also has an effect on the mixing, and varies with wind direction and strength. Thus the winds and tides act to mix and disperse waters internally, within the Firth.

As well as the mixing within the Firth, exchange between the Firth and the Hauraki Gulf occurs as the result of the strong mixing actions that the tides and winds induce between these waters. Estuarine exchange, driven by the balance between freshwater inflow and more saline ocean-affected waters from offshore, also causes mixing. Finally, advective 'eddy' events of offshore waters toward the Firth induce exchange between the Firth and the greater Gulf. Altogether, there is considerable scope for mixing and exchange, between the Firth and greater Hauraki Gulf, which affect Firth dynamics.

Exchanges have been estimated between the Hauraki Gulf (of which the Firth is a part) and the shelf waters immediately offshore (Zeldis and Smith 1999; see also the LOICZ website : http://data.ecology.su.se/MNODE/ New_Zealand/HaurakiGulf/ Haurakibud.html.). Hydrographic data (salinity) from NIWA surveys of the region conducted in 1996-97 were used for this (Figure 3). The authors estimated the amount of mixing that must have occurred, to produce the observed salinity contrasts between the Gulf and shelf, given the amount of freshwater inflow and evaporation (from NIWA hydrometric databases). The results showed that the mixing rate was 580 billion m^3 per year, and the mean residence time of water in the Gulf was 56 days. The main source of freshwater for the Gulf, the Waihou and Piako Rivers at the head of the Firth of Thames, contributed a total of 3 billion m^3 per year to this flux (it is this water which generates the estuarine flows in the Firth).

Figure 3

Shelf and Gulf stations sampled in Oct., Dec., 1996 and Jan./Feb. 1997, for the Hauraki Gulf nutrient budget model of Zeldis and Smith (1999). The Firth Stations were sampled in quarterly in 2000-2001, for incorporation into a future version of the budget (due to be completed in 2003/2004).
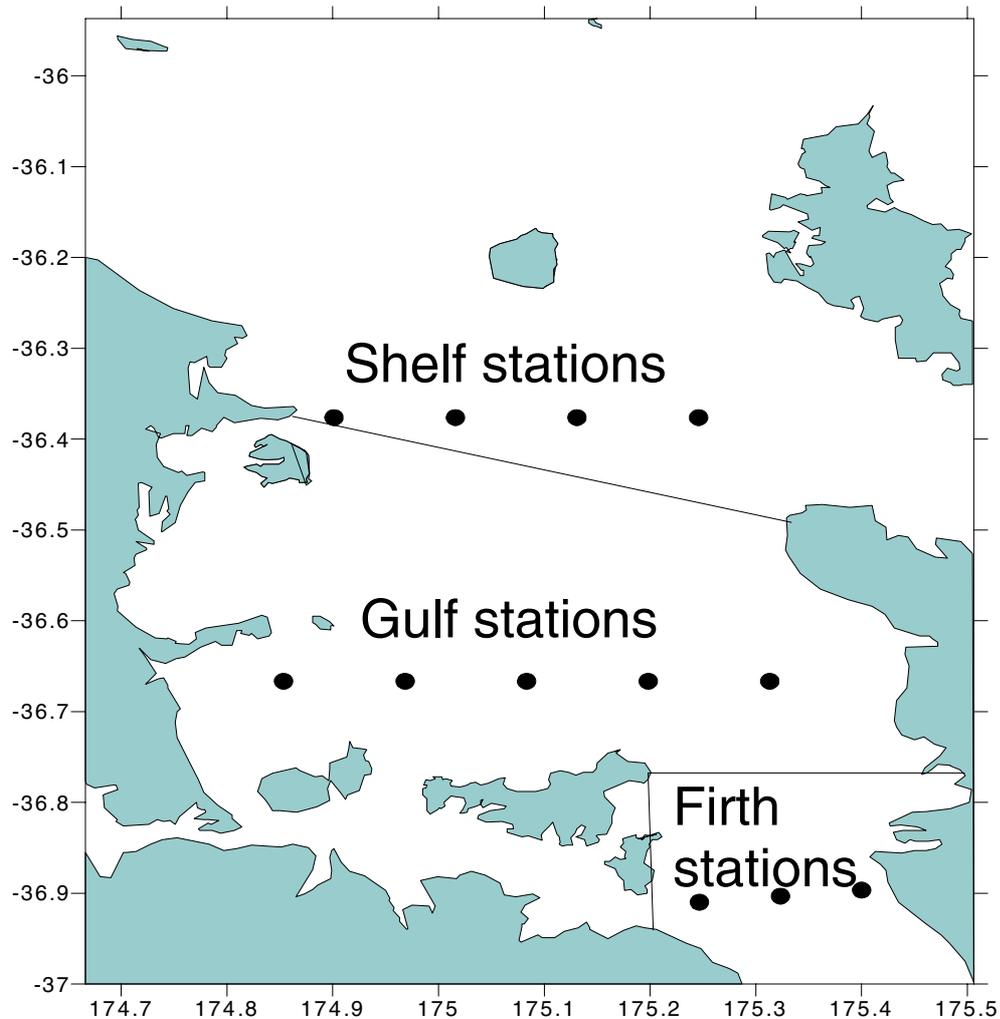
Nutrient concentrations were also sampled on the 1996-97 surveys, along with nutrient loads in rivers and sewage [Williams, 1998 #643; Wilson, 1998 #644; B. Vant, Environment Waikato *pers. comm.*; A. Smaill, Auckland Regional Council *pers. comm.*]. This made it possible to examine behaviour of nutrients with respect to the physical flows, to deduce their net uptake and release due to biological processes in the Gulf during an ENSO-neutral period (i.e. no strong El Niño / La Niña signal). It was found that the flux of dissolved inorganic phosphorus, ΔDIP, was positive (out of the Gulf, onto the shelf). This implies the Gulf was 'heterotrophic' overall (ie it remineralised (consumed) more organic matter than it produced through primary production). The phosphorus that is released by net-remineralisation is exported from the Gulf into the shelf waters. In

contrast, the flux of dissolved inorganic nitrogen, ΔDIN, was found to be negative, showing that the Gulf was a net sink for DIN. The mechanism for this nitrogen loss is 'denitrification' (the bacterially-mediated process by which nitrogen gas is produced from oxidised organic matter; there is no similar process by which phosphorus can be lost from the aquatic system). The DIN influx from sewage was about 15 % of the DIN influx from rivers (chiefly the Waihou and Piako rivers draining Waikato pastoral land). The summed DIN flux from sewage and rivers was about 25 % of the denitrification loss rate. Although the amount of water coming from rivers was small compared to mixing with shelf waters (see above), its nutrient concentrations were much higher, causing DIN inputs from rivers to be about 40 % of that derived from mixing with the shelf waters. Thus, the dissolved nutrient fluxes from sewage were substantially less than the fluxes from pastoral rivers in the catchment, while the riverine nutrient fluxes (mostly from the Firth) were somewhat less than half that of the fluxes due to mixing with the shelf waters offshore.

A recent NIWA research voyage (Nodder 2000), made in December 1999, surveyed sedimentary structures and biogeochemical functioning of Firth of Thames, greater Hauraki Gulf and shelf-slope sedimentary environments. Rates of denitrification and organic matter remineralisation were highest in sandy silt and silty clay sediments of the inner Hauraki Gulf, consistent with the budget results of Zeldis & Smith (1999). Results of a computer simulation study of sediment biogeochemical dynamics in Hauraki Gulf sediments (Giles 2001) also were consistent with this finding.

It must be emphasised that the budget of Zeldis & Smith (1999) was very simple, with only a 'Gulf' box and a 'Shelf' box. Because of its simple structure, the model averaged the fluxes for the greater Hauraki Gulf (including the Firth of Thames), relative to the shelf outside the Gulf. Thus, it could not resolve smaller-scale fluxes of water and nutrients between the Firth and the greater Gulf. NIWA is presently extending the budget by including a box for the Firth of Thames (Figure 3). Samples have been taken quarterly over 2000-01 (a La Niña period) in shelf, Hauraki Gulf and Firth of Thames boxes to explicitly determine these fluxes. This will allow us to determine the relative contributions of riverine nutrient input (largely Waikato pastoral) and oceanic nutrient input to the Firth (this exercise is scheduled for completion in 2003/2004). These results are expected to be significant to issues of Waikato catchment and Firth of Thames coastal management.

In conclusion, the greater Hauraki Gulf (incl. The Firth of Thames) is a net exporter of inorganic phosphorus. This implies that phytoplankton production within the Firth of Thames is inadequate to offset imports of organic phosphorus (e.g. from rivers and the continental shelf) – which are subsequently broken down into inorganic form within the Firth. A reduction is phytoplankton densities within the Firth can be expected to increase the quantity of inorganic phosphorus exported onto the continental shelf. This is, however unlikely to have any ecological consequences because phosphorus is not

the element which limits biological production on the shelf. In contrast, the greater Hauraki Gulf is a net importer of dissolved inorganic nitrogen. This import is a consequence of denitrification operating within the sea-floor sediments. Nitrogen is usually the limiting element within the Hauraki Gulf (incl. Firth of Thames). Aquaculture operations are known to influence the redox potential and organic content of the sediments below farms, and denitrification rates tend to increase below shellfish farms (Kaspar et al. 1985). Thus, if sufficiently extensive, shellfish aquaculture may substantially modify (reduce) nitrogen concentrations within the Firth of Thames. Given that nitrogen is often the element which limits primary production in this region, this would probably result in reduced rates of primary production, lower phytoplankton standing stocks, and perhaps also changes in the species composition of the phytoplankton community.

## 2.3 Physical processes and primary production in the Firth of Thames

Physical conditions strongly affect phytoplankton abundance by affecting rates of nutrient and light supply (Mann & Lazier 1991). Physical conditions, nutrient supply and productivity of the northeastern continental shelf region, Hauraki Gulf and Firth of Thames have been shown to vary from year to year because of large – scale processes originating offshore, driven by the El Niño / Southern Oscillation (ENSO) cycle (Zeldis, J.R.; et al. 2000; Zeldis, J.R.; et al. 2001a). Productivity in the region also varies seasonally, as the balance between light and nutrient limitation of phytoplankton growth changes through the year (Chang, F.H. et al. in review). Finally, productivity varies over the scales of days and weeks, as weather-driven events affect upwelling, nutrient supply and stratification (Zeldis, J.R.; et al. 1998). Below we describe in more detail this 'cascade' of scale-dependent processes, as they affect productivity in the Firth of Thames. We shall start at the larger, inter-annual scale.

2.3.1 Interannual variation: ENSO and the Firth

Field data (Sharples 1997; Sharples & Grieg 1998; Zeldis, J.R.; et al. 2000; Zeldis, J.R.; et al. in prep.; Zeldis, J.R.; et al. 2001a) and simulation results (Black et al. 2000; Proctor & Greig 1989) have demonstrated a strong link between northeastern New Zealand continental shelf hydrodynamics and the regional winds. Winds with a strong along-shelf component from the northwest cause upwelling, and those with southeast components cause downwelling (Figure 4). Satellite sea surface temperature pictures (Figure 4) show what happens to water temperatures on the northeastern shelf, under these differing conditions of upwelling and downwelling. The contrasting conditions are significant, because upwelled water is colder and richer in nitrogen and other nutrients. It fertilises the continental shelf, while downwelling conditions take the nutrient-rich water away from the coast and deplete the continental shelf of nutrients.

Figure 4.

Upper panels: This view up the coast to the north of the Hauraki Gulf shows two key wind driven current patterns – upwelling and downwelling. Winds are shown as feathered arrows, surface water is shaded light blue, and deepwater is shaded dark blue. The blue arrows depict water circulation. The orange symbols show the mooring sites near the Poor Knights Islands and in the Firth of Thames. Lower panels: These satellite images of sea surface temperature show that the cooling effects of upwelling (coloured green) in spring 1998 did not occur in spring 1999.
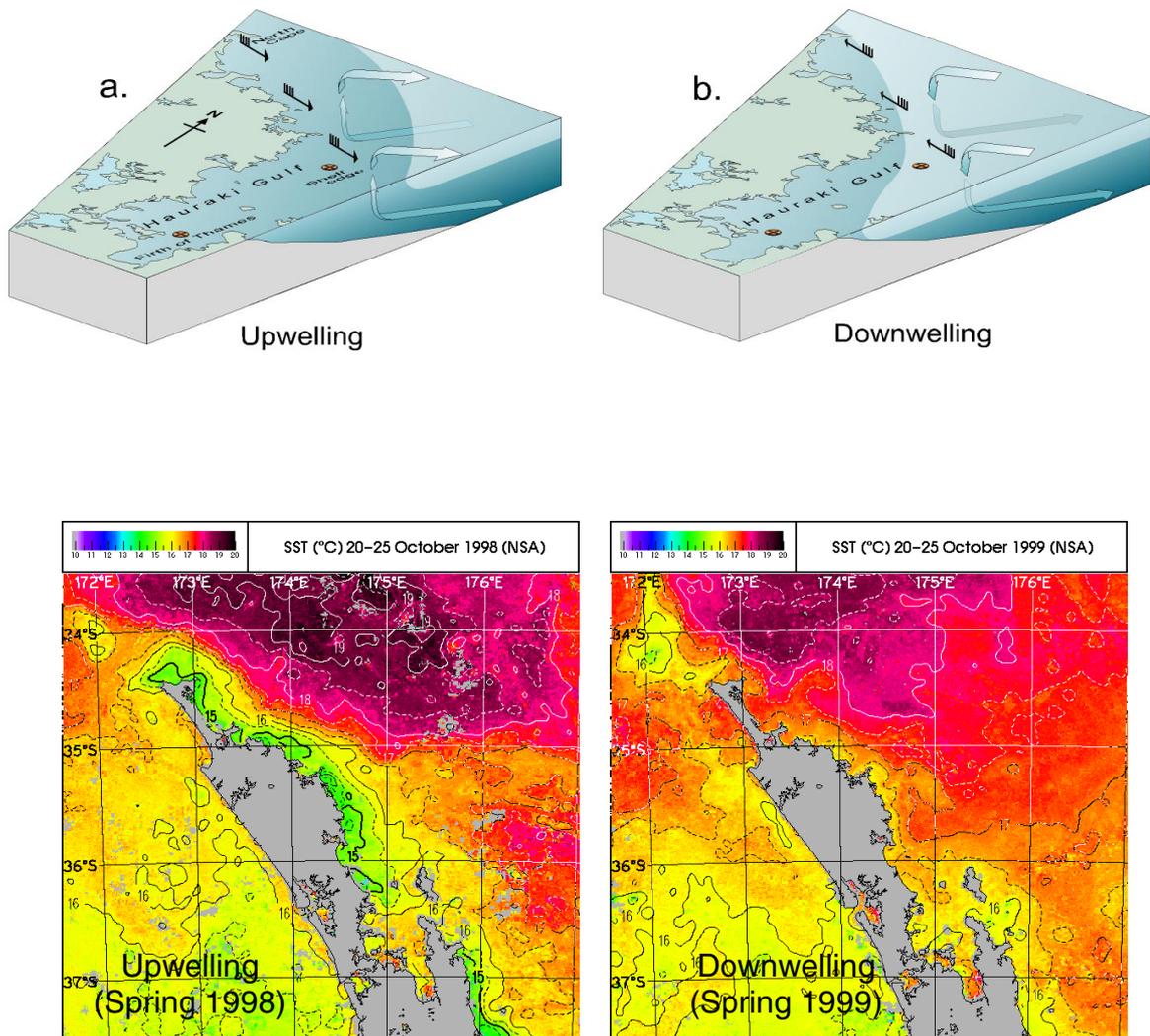
Wind data recorded at Mokohinau Island (Figure 5), at the entrance to the Hauraki Gulf, show that winds in the region are predominately from the southwest. These winds, however, have a significant along-shelf northwesterly component, especially in winter and spring. The other main winds are from the east, with strong along-shelf elements from the southeast. These are most common in summer. These seasonally variable winds cause a predominance of upwelling in winter and spring, but a change to downwelling in summer when nutrient poor, warm surface water is brought close to the coast (Sharples 1997; Sharples and Grieg 1998).

Figure 5.

Wind rose for Mokohinau Islands (data from January 1998 to July 2000), showing directions (bars) and wind speeds (km h^{-1}; shades).
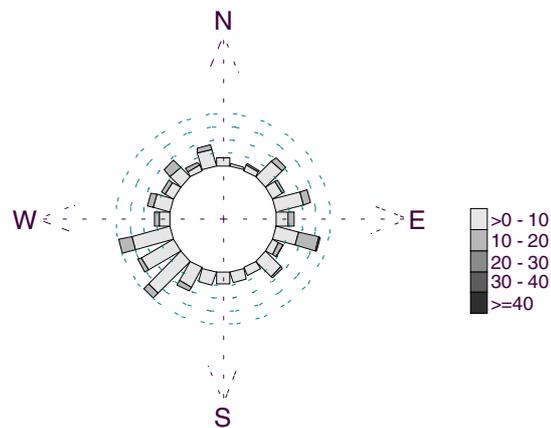
As well as this seasonal variation, there is important interannual variation in the prevalence of the alongshelf winds, related to the sign of the El Nino/Southern Oscillation index (Figure 6). El Niño periods (ENSO negative) typically bring westerlies. This is because, during El Niño, anticyclones are more common over the Tasman Sea and winds blow from the west. During La Niña though, the anticyclones are more common to the east of the country, which cause prevailing easterlies. Thus, there is a close relationship between the value of the ENSO index and the prevalence of upwelling and downwelling on the northeast shelf.

Figure 6.

El Niño/Southern Oscillation (ENSO) phases are closely related to wind patterns in the Hauraki Gulf region. Upper panel: During the mid-1990's strong El Niño and weak La Niña phases were most common. But since late 1998, La Niña has dominated ENSO. Lower panel: Northwesterly winds (indicated by data below the axis in the figure) were the dominant pattern up until late 1998. But since the start of dominant La Niña, southeasterly winds (above the axis), have been much more common, and stronger.
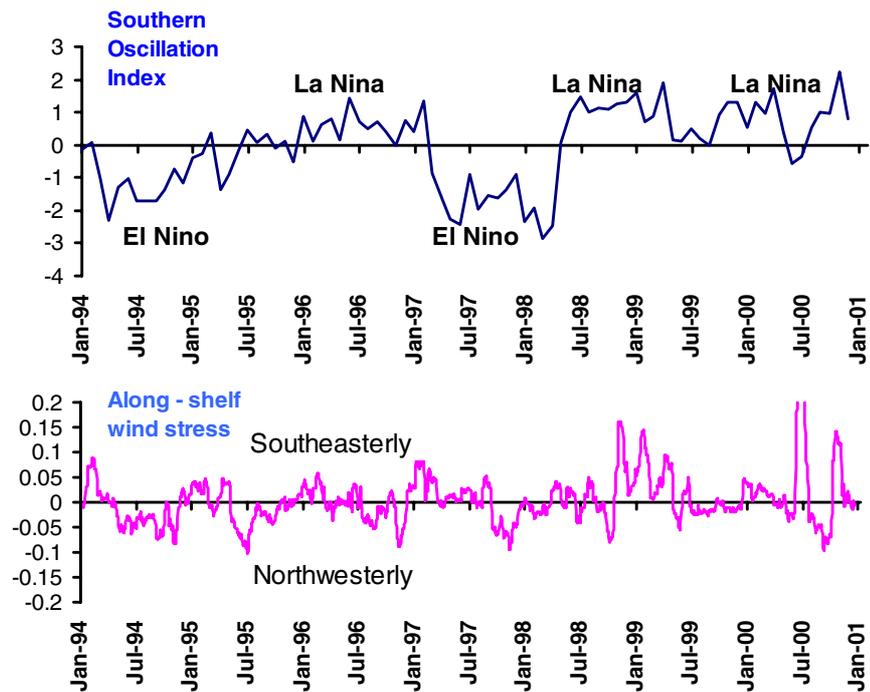
Events during 1998 to 2000 have provided a vivid example of these relationships, and how they affect the Firth of Thames. NIWA has been recording water temperature and salinity at the shelf edge since September 1998, using equipment moored near the Poor Knights Islands (Figure 4; a second mooring collects data in the northern part of the Firth of Thames). Upwelling affects not only the temperature of the water, but also the salinity. Deep water has low salinity, so when upwelling occurs the salinity on the continental shelf goes down. A salinity recorder on the mooring near the seabed shows that there was upwelling in late 1998 (Figure 7). Nitrate levels at the shelf edge near-bed were high as well (not shown). However, during most of 1999 there was a lot of downwelling, with high salinity, and lower nitrate levels. Since late 1999 and early 2000 there has been a mixture of upwelling and downwelling periods, as La Niña has weakened.

Figure 7.

Upper panel: Salinity measured at the Poor Knights increased in early 1999, when La Niña caused downwelling to start. This indicated that low-nitrate subtropical water was filling up the northeast continental shelf. In 2000 upwelling happened more often, when La Niña weakened. This indicated that nitrate was being replenished. Lower panel: In the Firth of Thames, as at the shelf edge, salinity increased in 1999. Nitrate levels became very low at that time. Then in 2000, lower salinity, upwelled waters returned, and nitrate levels increased.
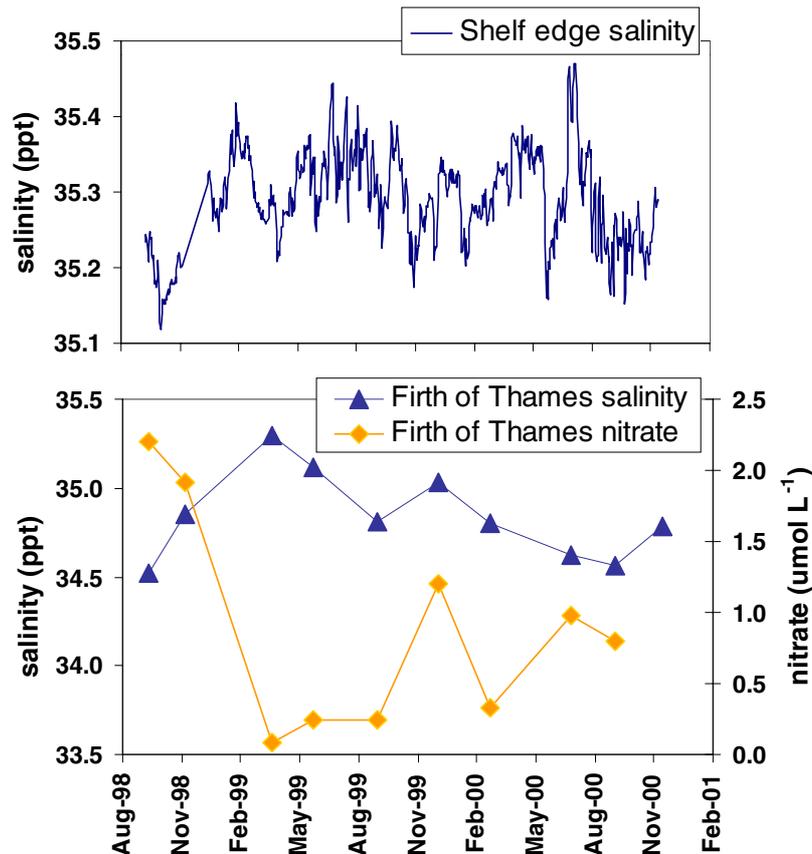
The waters on the shelf, with their varying nutrient loads, are introduced into the Hauraki Gulf by the mixing processes described in the previous section. Data from the second mooring, in the entrance to the Firth (Figure 4) make it possible to determine how this mixing influences the Firth. As at the Poor Knights site, water in the Firth in 1998 had low salinity and high nitrate (Figure 7), which indicated the presence of upwelled water. But through most of 1999, salinity was high and nitrate concentration was very low. In 2000, salinity started to drop, and nitrate increased again, as La Niña weakened.

NIWA has visited these two moorings every 3 months to download and maintain the instruments. On these voyages, oceanographic samples were taken at each mooring site and a number of other stations. The first two years of results for the Firth of Thames site are shown in Figure 8. The temperature record shows the typical seasonal

progression of temperatures, although the summer, autumn and winter of 1999 were considerably warmer than in 2000. The 3-monthly samples (Figure 8) appear to have faithfully captured the seasonal temperature signal: thermistors on the Firth mooring, recording temperature every 5 minutes at 6 depths in the water column, have given a very similar temperature record over the two years (Figure 9). The 3- monthly salinity record (Figure 8) in the Firth also shows a difference between the two years, with 1999 having considerably higher salinity throughout the water column. It was unlikely that the 1999/2000 salinity contrast was driven by between-year variation of river flows into the Firth. The record of Waihou River flows shows no significant difference between these years (Figure 10). It is most likely that the temperature and salinity patterns in the Firth reflected the La Niña-driven downwelling events of 1999, followed by the return to intermittant downwelling and upwelling in 2000, detected at the shelf edge.

Figure 8.

Depth by time contour plots of selected physical, chemical and biological parameters sampled 3-monthly at the Firth of Thames mooring site (see Figure 4 for site position) between Sep. 1998 and Sep. 2000, in the NIWA FRST_funded coastal exchanges programme. Dots indicate sample depths. The upper three panels (temperature, salinity and density differential) provide indications of the water-column's physical characteristics (notable degree of stratification). The panel entitled *ChlaT* indicates chlorophyll concentration in the >0.2 μm size fraction (i.e. this is a measure of the total phytoplankton abundance). *Phaeo*pigments are a breakdown product of chlorophyll and provide a measure of one part of the detritus in the system. NO_3-N, NH_4-N and urea are the three constituents of dissolved inorganic nitrogen. The ratio of NH_4-N to NO_3-N provides an indication of the extent to which the system is dependent upon nitrogen which has recently been regenerated from organic matter (i.e. the extent to which the system is dependent upon recycled nutrient rather than imported nutrient). DRP (dissolved reactive phosphorus) and DRSi (dissolved reactive silicon) are the two other elements which can sometimes limit phytoplankton growth in coastal waters (though they do not appear to do so in the Hauraki Gulf / Firth of Thames). The small map indicates the locations of the samples which have been used in generating these illustrations.
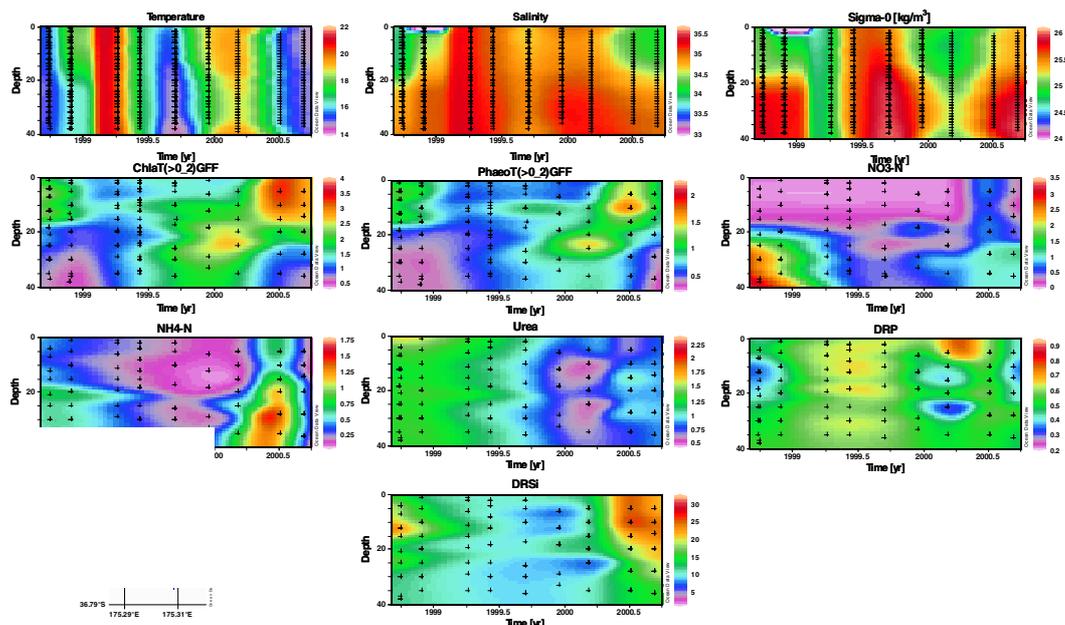
Figure 9.

Profile of temperature at the Firth of Thames mooring site, sampled every 5 minutes between Sep. 1998 and Nov. 2000. Horizontal lines indicate average depths of thermistors.
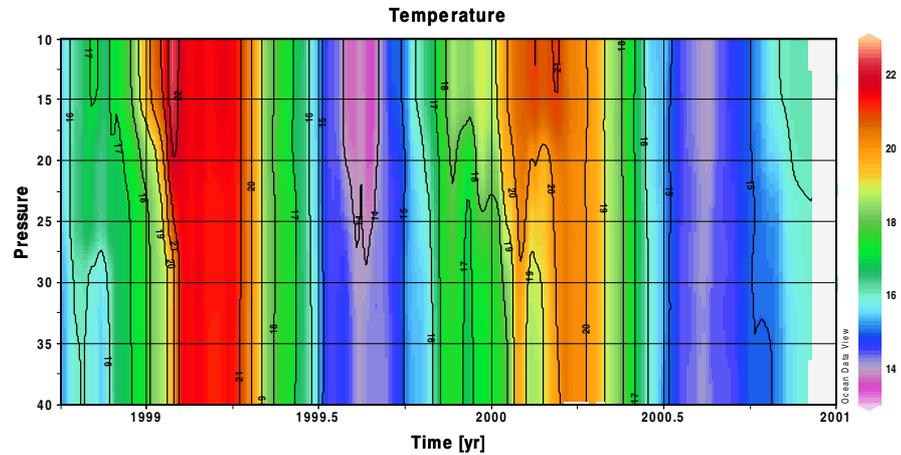
The key nutrient for coastal phytoplankton is nitrogen, usually as nitrate (NO_3^-) but also as ammonium (NH_4^+), because nitrogen is almost always the 'limiting nutrient' for phytoplankton growth. This means that when nutrients are depleted by phytoplankton growth to the point where growth stops, it is nitrogen which is depleted first. The 3-monthly profiles (Figure 8) show that starting in early 1999, NO_3^- and NH_4^+ nitrogen were driven to very low concentrations (<0.5 µmole L^{-1}). This concentration is considered to indicate nitrogen limitation of phytoplankton growth (Eppley et al. 1970). Starting in early 2000, both NO_3^- and NH_4^+ concentrations started to increase, in the lower water column. Although silicate (DRSi) concentration was lower in 1999 than in 1998 and 2000, at no time did it reach limiting concentrations (this is partly because it is supplied in abundance by river flow). Phosphorus (DRP) concentration showed a different and interesting pattern, and actually increased during the low nitrogen period of 1999. This indicated extreme nitrogen limitation, with the build up of DRP probably resulting from remineralisation of organic material in sediments combined with under-utilisation of DRP, driven by absence of nitrogen. At this time the ratio of nitrogen to phosphorus in the water column approached 1; in freshly upwelled waters it approaches 16 (Redfield 1934).

Figure 10.

Waihou flows. Flow volumes of the Waihou River at the head of the Firth of
Thames from Jan. 1997 to Feb. 2001. Data are from NIWA hydrometric data base.
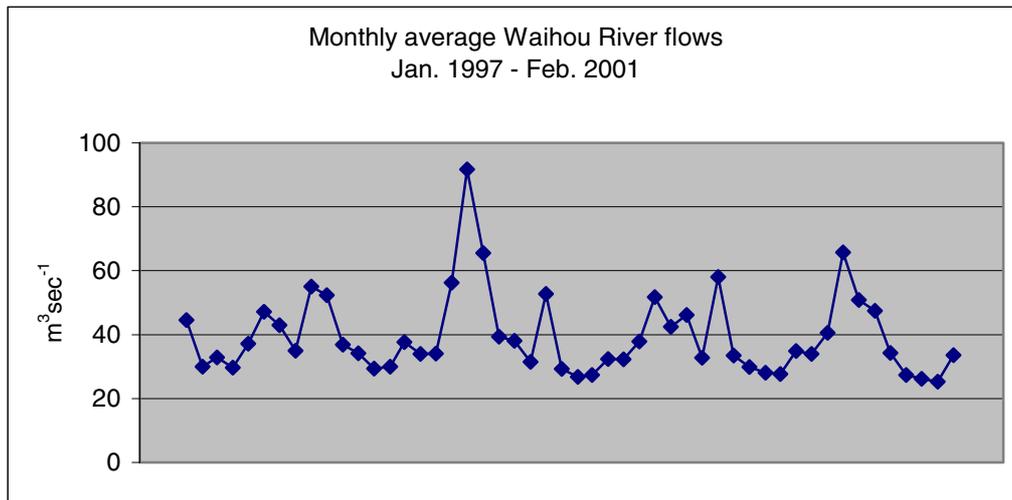

The interannual variation in nitrogen supply has driven similar variations in phytoplankton abundance. Figure 8 shows that in 1999, chlorophyll-*a* (chl-*a*) concentrations, especially in the upper half of the water column, were much lower than at comparable times in 2000. This was especially true during the spring bloom period in August – October. The spring bloom in 1998 did not appear strong, even though nitrate was abundant that year. This unexpected observation suggests that it is again worth asking, whether 3-monthly sampling was sufficient to resolve the potentially rapid changes phytoplankton assemblages can undergo. To answer this we are using two additional data sets. The first is derived from a collaboration with an oyster farming company in Coromandel, Pacific Marine Farms. They started taking water samples every two weeks at the entrance to Coromandel Harbour, for analysis by NIWA, in July 1999. These fortnightly results (see Zeldis (2001b)) revealed the same pattern as in the central Firth, with nitrate and chl-*a* increasing and salinity decreasing between spring 1999 and spring 2000. These more frequent data indicated that the contrasts between 1999 and 2000 in chl-*a* were observed accurately by 3-monthly sampling in the central Firth.

The second data set comes from SeaWiFs satellite ocean colour determinations of chl-*a* concentration. . It is known that sediments and dissolved organic materials from rivers bias satellite chl-*a* determinations in coastal regions. However, the Waihou River flow record (Figure 10) does not suggest that spring flows varied significantly across this time series, with flows averaging about 40 cumecs during Aug.-Oct. each year. This suggests that river flow variations would not have driven large interannual variation in satellite-based chl-*a* determinations. The SeaWiFs data have been accumulated daily, and are

summarised as monthly means in Figure 11. The data show that the spring bloom of 1998 was in fact strong, which was not reflected by the single profile acquired in the 3 monthly sampling in spring 1998.

Figure 11.

Satellite maps of surface ocean chlorophyll show that spring 1999 was a poor one for phytoplankton in the northeast region and Hauraki Gulf. Concentrations range from near zero (blue) to over 3 mg per cubic metre (orange). Black areas are cloud-covered. Right panels: Hundreds of satellite images like the ones on the left were put together to get a summary of chlorophyll every month since 1997. Spring of 1999 had a very poor spring bloom, when upwelling stopped. The boxes in the spring 1997 picture (left panels) show the areas of the northeast shelf and the Gulf that are summarised in the right panels.
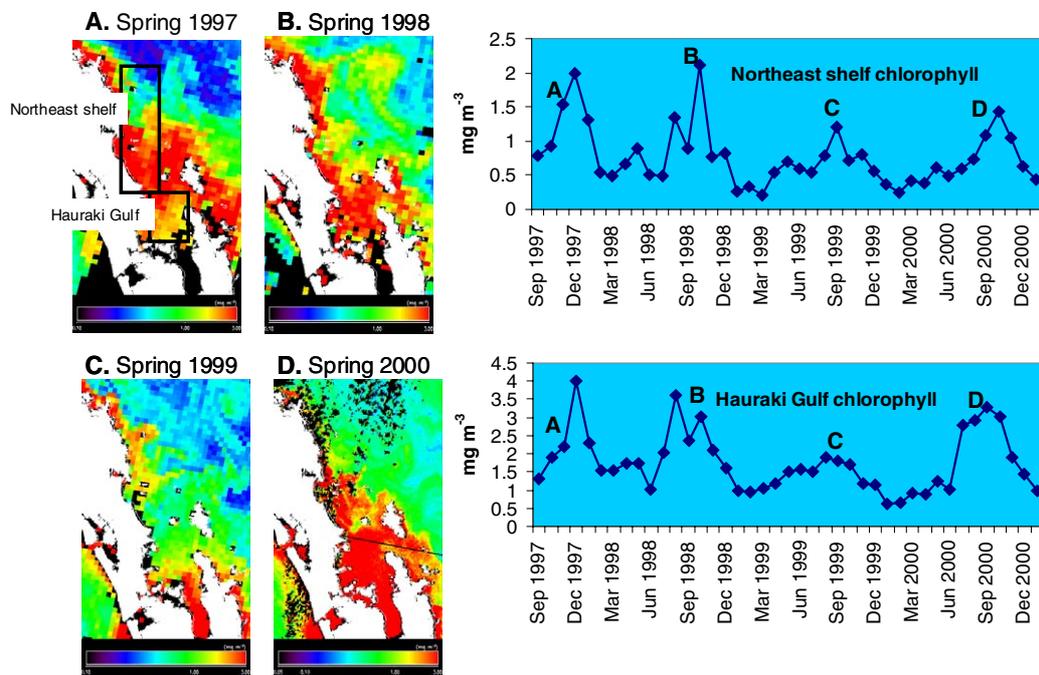
Courtesy NIWA Ocean Colour Objective

Most significant, however, is the very weak spring bloom observed in SeaWiFs data in 1999, compared to the other 3 years of the time-series (Figure 11). This was consistent with the general trends throughout all the data sets, signifying the oligotrophic (low nutrient and chl-*a*) character of the La Niña-affected 1999 year, and mesotrophic (moderate nutrients and chl-*a*) character of years where upwelling was more prevalent. As such, fluctuating nitrate levels, driven by large scale physical processes originating well offshore, clearly impact Firth productivity.

2.3.2 Variation in productivity at seasonal to daily scales: stratification, nutrients and light in the Firth

Next we consider processes at shorter time scales: those operating seasonally, weekly and daily. It is at the shortest scales that phytoplankton cells react to nutrient and light conditions and it is the integration of these short-term events which determines the magnitude of production and its seasonal and inter-annual variations. Production is strongly affected by stratification (density layering) of the water column. Strongly stratified water columns restrict nutrient supply to the upper part of the column from below, preventing nutrients from reaching shallower depths where light is sufficient to support net phytoplankton production. This restriction is most acute in summer, when phytoplankton deplete upper–layer nutrients, and strong thermal stratification prevents vertical mixing of new nutrients. Alternatively, if the water column is strongly mixed, phytoplankton may receive abundant nutrients from deep in the column, but be starved of light, by becoming deeply mixed and spending too much time in the dark. This is most acute in winter, when light is low, and when vertical mixing can be strong due to high winds and lack of thermal stratification. Situations intermediate between these two extremes tend to support highest productivity (Mann & Lazier 1991). The spring bloom is the most obvious example, when stratification is intermediate, and the water column has been restocked with nutrients by overwinter regeneration and vertical mixing. But high production events also may occur in summer, if stratified periods alternate with mixing from wind events; in autumn, when thermal stratification starts to break down, injecting a pulse of nutrients into higher light surface waters; and in winter, when high river flows may stratify the water column, trapping riverine nutrients in the surface waters.

All of these processes have been observed in Hauraki Gulf and Firth of Thames studies. The seasonal cycle of the spring bloom followed by the summer and autumn declines of chl-*a* is obvious in the SeaWiFs data (Figure 11), over the northeastern shelf and Hauraki Gulf region. This has been well documented for the northeast shelf and Hauraki Gulf by (Chang, F.H. et al. in review), using data collected in 1996-97. Assessments of chl-*a* were also made across the Gulf between November and January in 1985-86, 1986-87 and 1987-88, during studies of zooplankton, fish eggs and larvae by MAF Fisheries Research (Zeldis, J. et al. in review; Zeldis, J.R. 1992; Zeldis, J.R.; et al. 1995). Within the Firth, chl-*a* concentrations in the upper 15 m of the water column show the spring to summer decline (Figure 12).

Water column mean concentrations were between 3 and 4 µg L^{-1} in Nov. (late spring) and declined to 1-2 µg L^{-1} by Jan. (summer). Quite high concentrations (4 – 5 µg L^{-1}) were detected at the two sites furthest south in the Firth. These chl-*a* levels can be compared with another well-studied coastal region in New Zealand: Pelorus Sound (Table 3), which has been studied in other NIWA research programmes. Mean and

median chlorophyll concentrations within the Firth of Thames are nearly twice those measured in the Marlborough Sounds – suggesting that the Firth of Thames may yield higher shellfish growth rates than the Marlborough Sounds.

Figure 12.
(A) Site map of Firth of Thames showing positions of MAF sampling sites (Northwest, Northeast, Southwest and Southeast) occupied in 1986-88. (B) Chlorophyll a (chl-a) concentraion in Firth of Thames: spring and summer samples from MAF surveys in 1986/87 and 1987/88 taken between 0 and 15 m. (Source: NIWA, unpublished data).
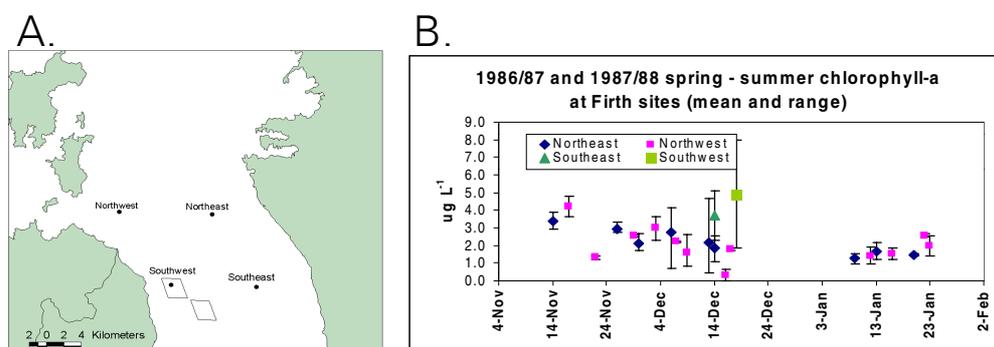
The spatial distribution of chl-a over the entire Hauraki Gulf and Firth from the MAF Fisheries research surveys is shown in Figure 13. This plot shows average chl-a concentration in the upper 30 m of the water column at each station (dots; a total of 850 chl-a determinations) across the entire time series, and so removes the late spring to summer seasonal signal. It shows that the Firth sustains the highest chl-a concentrations among all sampled coastal areas of the Hauraki Gulf (Tamaki Strait is excluded because of lack of samples). Why does the Firth appear to be so productive?

Table 3

Comparison of phytoplankton biomass (as measured by chlorophyll-*a* concentration) in Firth of Thames and the Marlborough Sounds.

Locations	Dates	No. of values	Range, mean, median (µg chl-*a* L)
Firth of Thames (all sites in Figure 12)	November to January 1986/87 and 1987/88; quarterly 1997-2000	33	0.3-4.9, mean = 2.2, median = 1.9
Beatrix West	August 1995 – April 1999	186	0.3-4.5, mean = 1.4, median = 1.1
Beatrix East	August 1995 – April 1999	187	0.2-11.2, mean = 1.4, median = 1.0
Tawero Point	September 1997 – April 1999	126	0.3-4.7, mean = 1.5, median = 1.3

[1] Unpublished NIWA data: 'No. of values' is number of 0 - 15 m profiles used, typical number of depths assayed per profile is 3.

[2] Ross et al. (1998a; 1998b). 'No. of values' is no. of weekly 0-15 m integrated water samples collected by the Marlborough Sounds Shellfish Quality Programme.

Figure 13.

The spatial distribution of chl-*a* over the Hauraki Gulf and Firth of Thames from the MAF
Fisheries Research surveys from November to January in 1985-86, 1986-87 and 1987-88.
The plot shows average chlorophyll-a concentration in the upper 30 m of the water column
at each station (dots). Tamaki Strait is excluded because of lack of samples.   (Source:
NIWA, unpublished data).
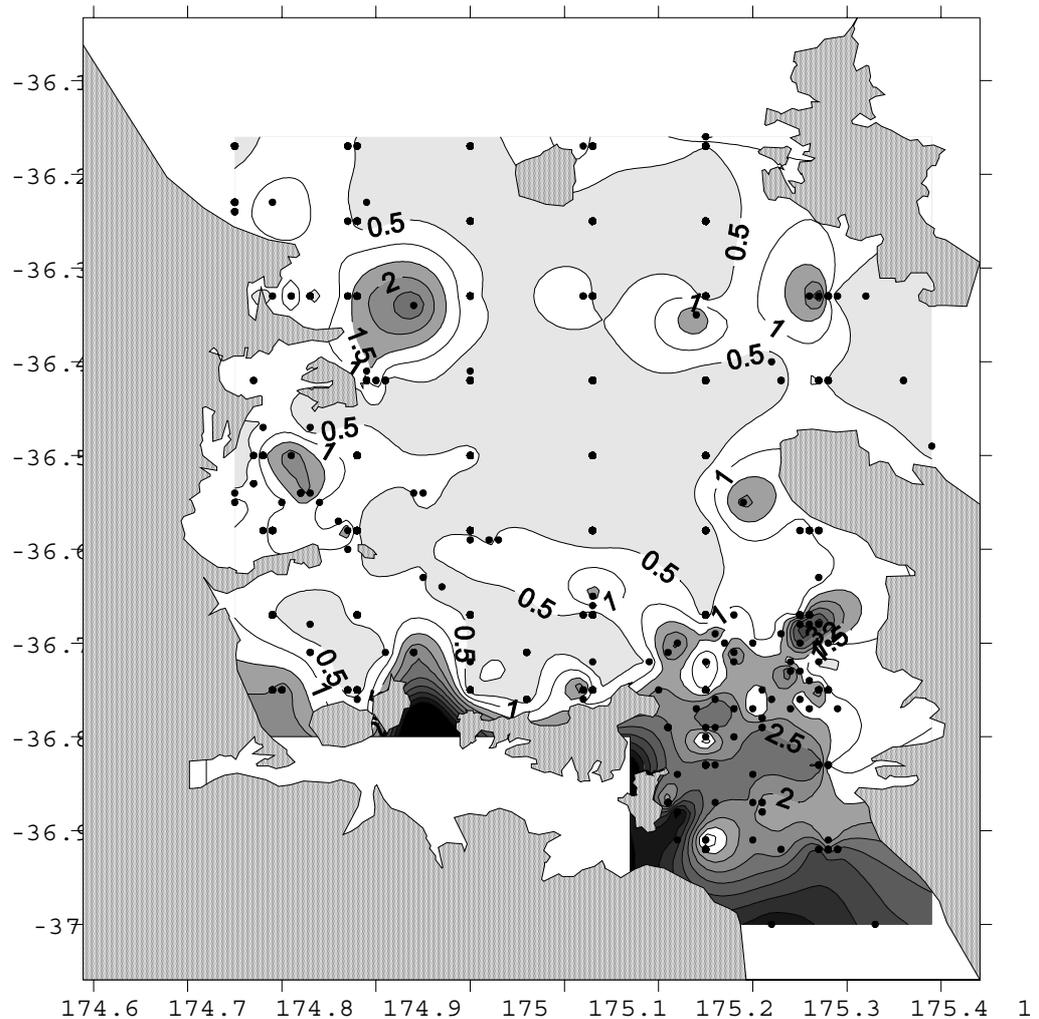
As mentioned above, the dynamics of stratification may be important. We examined
water column physical data from MAF and C-SEX surveys for the Northwest site (Figure
12), to determine stratification conditions (Table 4). During late spring and summer, the
water column appears to be alternately mixed or stratified over weekly time scales,
most likely because forces driving stratification are weak (i.e., there are only small
changes in temperature or salinity vertically down the water column). Stresses from tidal
bottom friction will work against these weak buoyancy forces, breaking down
stratification. Indeed, a numerical circulation model of the Hauraki Gulf (Grieg & Proctor

1998), shows that the tidal bottom stresses in the coastal Firth, for example, in the vicinity of the Northwest site, are near those sufficient to consistently break down seasonal (i.e., thermal) stratification. The consequent injection of nutrient rich, 'sea-floor' water into the surface may be the reason why the mouth of the Firth sustains higher chlorophyll concentrations than the Hauraki Gulf (Figure 13).

Table 4.

Hydrographic conditions at Northwestern site, Firth of Thames (see Figure 12) . Shown for each date are water column depth (m from the sea surface), the change (Δ) in temperature (^{o}C), salinity (psu), and density (kg m^{-3}) per meter increase in depth, the stratification condition of the water column ('stratified' if density increases more than 0.03 kg m^{-3} over the water column between 2 m depth and the bottom, otherwise mixed), and the depth of mixing (m).

Date	Study	Station	Depth	ΔTemp (m^{-})	ΔSalinity (m^{-})	ΔDensity (m^{-})	Condition	Mixed depth
Nov 1986	MAF	95	15				mixed	bottom
Dec 1986	MAF	17	16				stratified	8
Jan 1987	MAF	9	12				mixed	bottom
Jan 1987	MAF	80	12				stratified	8
Nov 1987	MAF	68	12				stratified	9
Dec 1987	MAF	111	14				mixed	bottom
Dec 1987	MAF	9	14				stratified	8
Jan 1988	MAF	63	12				stratified	10
Apr 1999	C-SEX	3	14				mixed	bottom
Jun 1999	C-SEX	10	14				stratified	10
Sep 1999	C-SEX	4	14				stratified	4
Jul 1999	C-SEX	5	16				stratified	9
Sep 1999	C-SEX	20	14				stratified	2

In contrast, in winter and early spring, vertical changes in salinity are large, and the water column becomes moderately stratified (Table 4; see also the salinity and Sigma-0 contours in Figure 8. This is driven by freshwater runoff which originates from the Waihou and Piako Rivers, by far the largest rivers in the area. The addition of buoyancy due to freshwater in winter and early spring apparently allows stratification to resist tidally – driven vertical stirring at this time.

Along with stratification, it is likely that mixing and advection of offshore waters into the Firth plays a pivotal role in sustaining its productivity. A vivid example of this interaction of advection and stratification on light and nutrient supply is shown in Figure 14. The instruments were a fluorometer, (measuring chl-*a* and primary production), an *in-situ* nitrate analyser (measuring nitrate at 30 m depth), an Aquamonitor (measuring nitrate,

ammonium, and phytoplankton cells at 10 m depth), and thermistors attached along the mooring wire between 10 m and the bottom (measuring temperature). The thermistor record shows that the water column initially was weakly stratified. Then, later in October and through November, pulses of cool water advected into the Firth in the lower water column, bringing in high concentrations of nitrate (this was associated with upwelling detected by the shelf edge mooring: see Figure 7). Although increased productivity was associated with the nitrate pulses and the weak stratification, not until the water column became strongly stabilised by summer warming in mid-December did a dramatic increase in productivity occur. The nitrate was depleted by this latter event, and eventually ammonium (the waste product of secondary production) became more common.

Phytoplankton cells were also captured and preserved by the Aquamonitor sampler during this time series (Figure 15), with a sample acquired every 4 days. Diatom cells were associated with the weakly stratified, deep pulses of high nutrient water, but these subsided as stratification intensified in early December Flagellated cells (dinoflagellates mainly) were the main components of the stratified bloom. When nutrients were depleted later in summer, the smallest flagellated cells, the nanoflagellates, became more common. This is very similar to the seasonal progression of phytoplankton taxa documented for the greater Hauraki Gulf by Chang *et al.* (in review), from the NIWA surveys conducted in 1996-97 (See also Maddock & Taylor (1984) for an earlier study made in Jellicoe Channel). It should be noted that the very smallest size class of phytoplankton, the picophytoplankton, were not sampled by the Aquamonitor because the Aquamonitor technique does not preserve them. Chang *et al.* (in review), show however, that these small cells become important in Hauraki Gulf coastal areas in late spring and summer, contributing approximately half of the phytoplankton biomass.

Figure 14.

Time series of parameters measured at the Firth of Thames mooring Sep. to Jun. 1998-99. Temperature data (coloured contours) are vertically resolved. Productivity (fluorometer) and nutrient (nitrate and ammonium) were collected at 13 m depth by INF fluorometer and Aquamonitor, respectively. Nitrate was also sampled at 30 m, by NAS2 in-situ analyser.
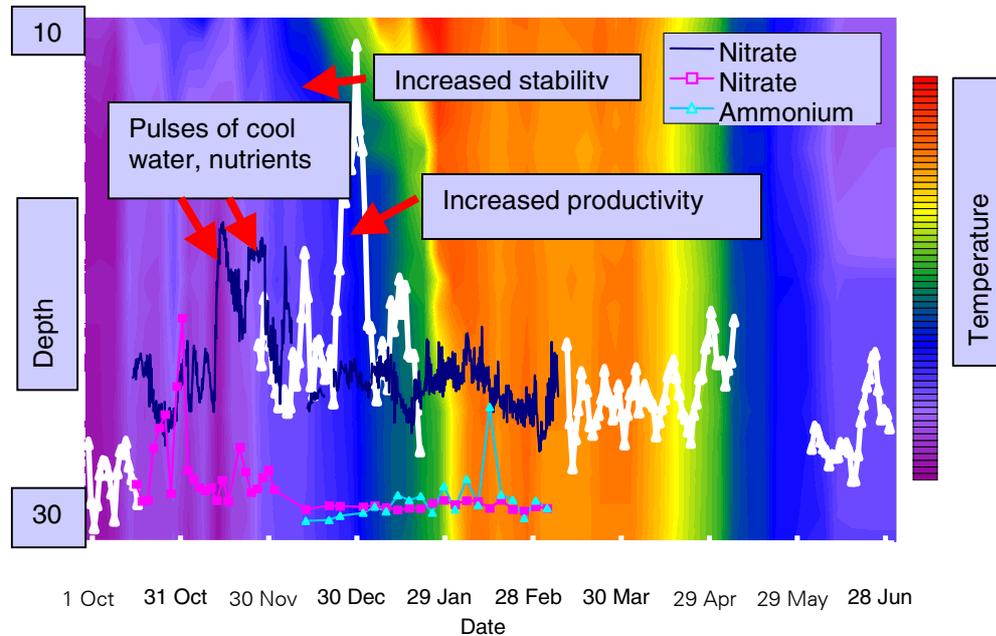

Figure 15

Biomass (as carbon) within 3 phytoplankton taxonomic groups sampled by the Aquamonitor instrument moored at 13 m depth on the Firth of Thames mooring, September 1998 to March 1999. (blue: diatoms, pink: dinoflagelaltes, orange: other (usually smaller) taxa).
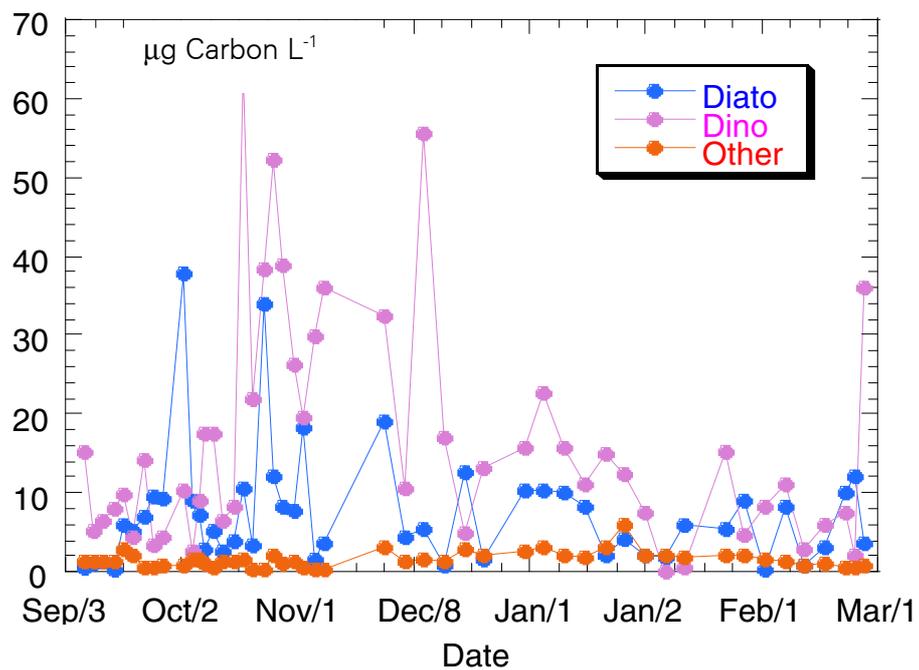
The incidence of toxic algal blooms (mainly of dinoflagellates) has also become a significant aspect of Firth of Thames ecology since the early 1990's, when outbreaks were first observed in the Hauraki Gulf (Chang, F.H.; et al. 1995). These outbreaks are a product of the stratification, nutrient and light climate in the Gulf described in Chang et al. (submitted), and are probably related to the summer subtropical intrusion events described previously (Chang, F.H.; et al. 1998).

These new data and published studies for the Firth illustrate how stability, light and nutrients affect production in the coastal environment. Overall, physical conditions in the Firth appear favourable for relatively high levels of production, with intermittent mixing and stratification in late spring and summer, and more consistent stratification in winter. This may allow phytoplankton biomass to accumulate in the upper water column more consistently than in areas further out into the Hauraki Gulf, where tidal stirring is weaker because of greater depth, and where there is less surface dilution from river inputs. Ultimately, however, the maximum levels of phytoplankton biomass will be limited by nitrate supply, which is highly variable between seasons and years, related to large-scale, offshore processes such as wind-driven upwelling and ENSO. The Firth clearly is affected by these processes, no doubt accounting for much of its productive capacity.

The productive conditions we have observed in the Firth suggest that it is a key area for secondary production in the Hauraki region. We examine this proposition in the next section.

2.4   Secondary production in the Firth

2.4.1   Zooplankton

Zooplankton communities of the Hauraki Gulf were first described by Jillet (1971), with regular sampling at an inner- and an outer Gulf site (Waitemata Harbour and Jellicoe Channel, respectively). Jillett's work provides detailed descriptions of the seasonal abundance and biogeographic affiliations of the mesozooplankton (>200 micron) for the region. Further surveys for microzooplankton (>5 microns) (Hall et al. in prep.) and mesozooplankton (Zeldis, J.R.; & Richards in prep) were conducted by NIWA during spring to summer 1996-97 on transects across the shelf and within the Hauraki Gulf. The inner Gulf transects from these studies will provide species and abundance data highly relevant to Firth ecology. Finally, the C-SEX time series (Section 2.3) has collected microzooplankton and mesozooplankton at 3 monthly intervals since September 1998 at the Firth mooring site, but these are still in a relatively early stage of analysis.

Some aspects of the zooplankton results from the 1985-1988 MAF Fisheries Research surveys (Section 2.3) were described by Zeldis *et al.*(1995). Here we present data (Figure 16) showing the total biomass of zooplankton, (excluding gelatinous species), captured

by plankton net samples (100 micron mesh) during these surveys. In the figure the data are summarised by each month of each year. The net samples were taken through the entire water column at each station, over a grid of 30 stations distributed across the region. The grids was usually occupied twice in each month. The exception was the first year, when the grid was occupied once per month, and when the Firth of Thames was not sampled. These zooplankton were largely the larvae and adults of crustacean copepods (oar-footed bugs) and cladocera (water fleas), but also include the larvae of benthic fauna such as crabs, barnacles and worms.

Zeldis *et al.* (in review) review these data. Three general patterns are important to note. First, zooplankton biomass increases from November to January, as animal populations grow by utilising the primary production of the spring and early summer. Second, there is interannual variability (correlated with ENSO-driven upwelling differences): note that the 3rd year (1987-88) had considerably higher biomass than the first 2 years. Third, the Firth of Thames frequently supports the highest zooplankton biomass of any region within the Hauraki Gulf. The last observation is consistent with the chl-*a* results (Figure 13), and the proposition that the southern Hauraki Gulf / Firth of Thames are very important in greater Hauraki Gulf regional secondary production.

Figure 16.

The biomasses (μg nitrogen L^{-1}) of zooplankton in the Hauraki Gulf region. The labelled line contours show summed biomasses of nauplii and copepodites (the larvae and juveniles of copepods) captured by the 100 μm net. The shaded contours show the biomasses of all other zooplankton (largely the larvae of benthic invertebrates). The values are water column means for all station grids (see Figure 13) occupied in each month, converted to natural logarithms. The shaded contours increase in intervals of 0.5, ranging from 0.5 to 4.5 (2 to 90 μg nitrogen L^{-1} on an arithmetic scale). To further reference the contours to arithmetic values, the antilogarithms of 1 through 5 are approximately: 3, 7, 20, 55, and 148. (Source, NIWA, unpublished data).
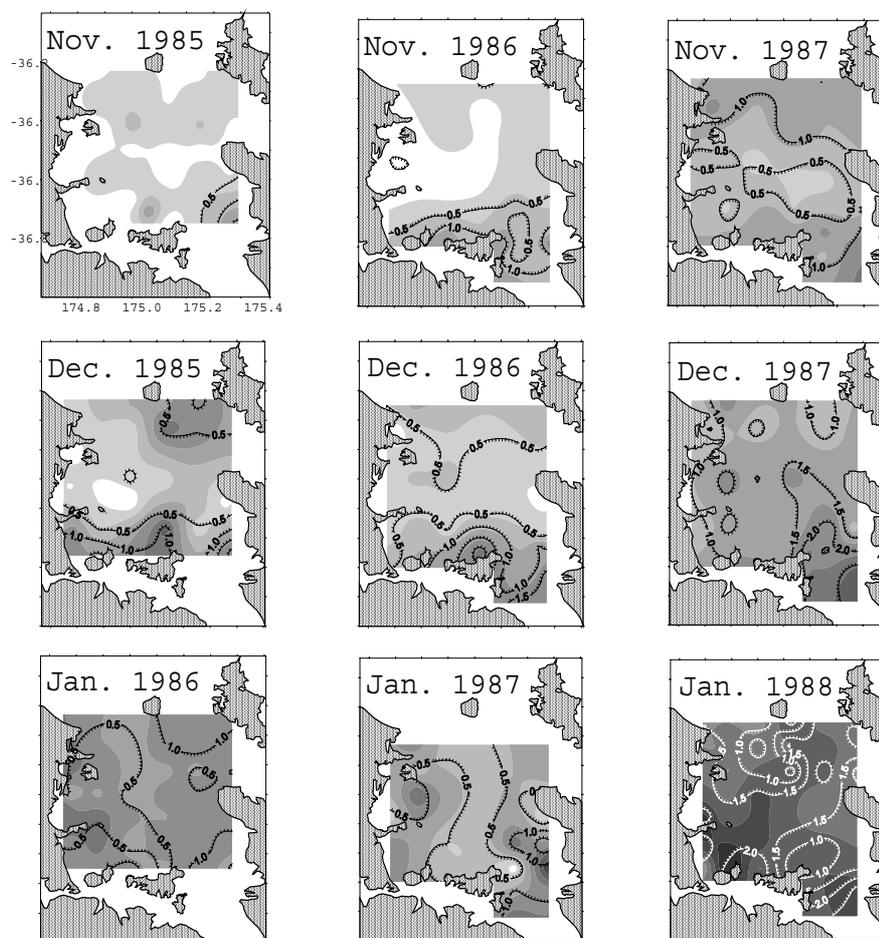
2.4.2 Fish (ichthyofauna)

Some of the earliest studies of the ichthyofauna of the Firth of Thames were conducted by Colman (Colman 1974 and citations within), who investigated movement, maturity, spawning, fecundity, and growth of flounder. NIWA and its MAF predecessors conducted an extensive time series of trawl surveys (to estimate the abundance of adult fish) throughout the Hauraki Gulf region between 1964 and 1997. This is described by Kendrick & Francis (in review), who plot catch distributions for a number of commercially important species; these show that the Firth of Thames is important particularly for

snapper and John Dory. The Firth also produced relatively high catch rates for rig, school shark, and barracouta juveniles (M. Francis NIWA, *pers. comm.*).

The ichthyoplankton (fish larvae) of the Hauraki Gulf region were first described in detail by Crossland (Crossland 1980; Crossland 1981). These results, while being a comprehensive account for the greater Gulf, did not include samples within the Firth of Thames. The MAF Fisheries Research surveys in 1985-88 (Zeldis, J. et al. in review) did include stations as far south as the central Firth. These surveys were geared toward understanding seasonal and interannual variability in the abundance of egg and larval snapper. Snapper are serially spawning fish, with each female producing a batch of planktonic eggs nearly every day, over the November to January spawning season (Scott et al. 1993; Zeldis, J.R; & Francis 1998). Figure 17 shows abundances of egg and larval snapper from net tows (365 micron mesh) taken over the same geographic areas as the zooplankton samples described in the section 2.4.1 (in this case, however, approximately 60 samples were taken in each grid in each month). The distribution of snapper eggs shows that spawning is located in coastal areas of the Hauraki Gulf, largely within the 30 m depth contour. Although there was little variability in snapper egg abundance between the 3 years, snapper larval abundance was much greater in the 3rd year. These dynamics were similar to those of the zooplankton community, probably because the larvae of snapper prey upon many of the these zooplankton species. Eggs and larvae of a number of other ichthyoplankton species (jack mackerel, blue mackerel, stargazer, ahuru,speckled sole, crested flounder, triplefin, opalfish, anchovy, not shown) showed similar dynamics. Finally, the Firth of Thames was consistently the most important site for snapper spawning and larval survival of any area in the Gulf, and was also where larval survival was highest (Zeldis, J. et al. in review).

Figure 17.

The abundance of snapper eggs (m^{-3}: labelled line contours) and of snapper larvae ((100 m)$^{-3}$: shaded contours) in the Hauraki Gulf. The values used are water column means for all grids occupied in each month, converted to natural logarithms. The shaded contours increase in intervals of 0.5 ranging from 0.5 to 8 (2 to 2980 on an arithmetic scale). To further reference the contours to arithmetic values, the antilogarithms of 1 through 8 are approximately: 3, 7, 20, 55, 148, 403, 1096, and 2980. (Source, NIWA, unpublished data).
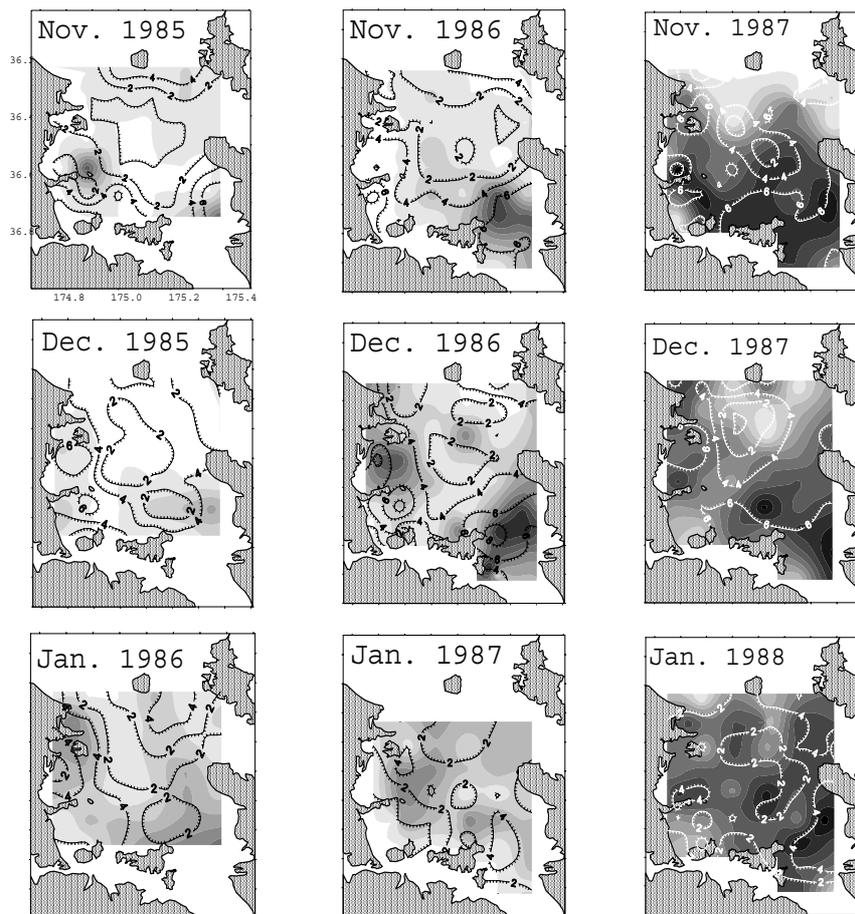
Another data set illustrating the importance of the Firth of Thames for snapper ecology is shown in Figure 18. These samples were taken during a 'Daily Egg Production Survey (DEPM) survey of Hauraki Gulf snapper made in November -December 1992 (Zeldis and Francis 1998). The purpose of the DEPM was to estimate adult snapper biomass for stock assessment. This estimate was made by dividing estimates of the abundance of newly spawned planktonic eggs (corrected for egg mortality), by the product of adult female spawning frequency, daily fecundity and stock sex ratio, determined using a coincident trawl survey. Figure 18A shows the abundance of newly spawned eggs over the survey area (from 300 plankton stations sampled in the 10 day survey). This shows the coastal distribution of snapper spawning, and also shows clearly that the Firth was

producing more spawning than any other coastal region in the Gulf. Figure 18B shows that adult snapper were very abundant in the Firth, and Figure 18C shows that, in the peak spawning period (Nov.- Dec.), most of these fish were spawning. As such, the Firth is clearly a crucial snapper spawning region within the Hauraki Gulf, which itself sustains the largest snapper fishery in New Zealand.

Figure 18.

A) Snapper egg abundance (proportional to circle size) from the DEPM survey for 0-6 hour-old eggs. Largest circle = 888 eggs m^{-2}. (B) Adult, mature snapper catch rate (kg km^{-1} trawled). Largest circle = 553 kg km^{-1} (C) Proportion of mature females spawning day^{-1}. Largest circle = 1. Crosses indicate zero catch or proportion. Survey strata are shown by polygons. (Source, NIWA, unpublished data).
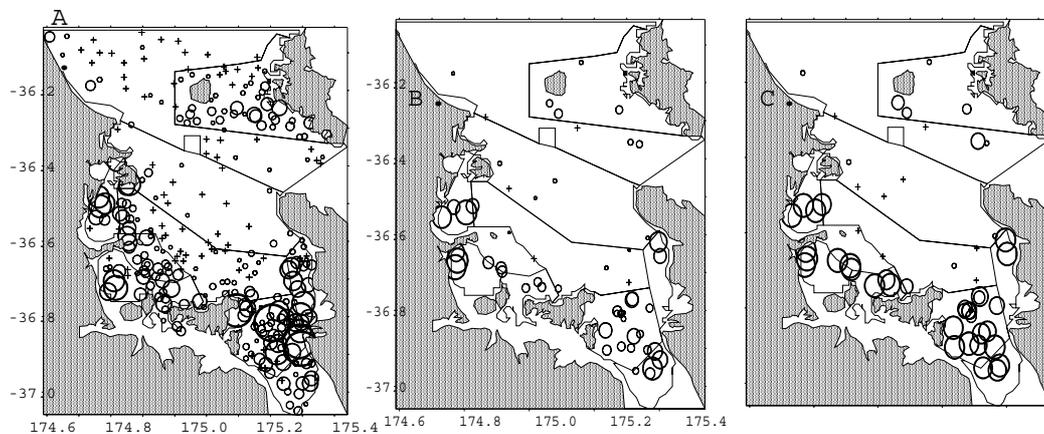
2.4.3 Benthos

The present nature of the benthic infauna and epifauna of the Firth are less well described than planktonic biota, but are the subject of current investigations by NIWA (Nodder 2001, FRST program contract CO1X0027) and as part of the Wilson Bay monitoring programme recently initiated by NIWA and the Group A Consortium (Zeldis et al. 2001). The Wilson's Bay program was put in place as part of the monitoring required as part of the Resource Consent associated with mussel farm developments at Wilson's Bay. It is designed to detect changes in the characteristics of seafloor and associated fauna immediately below the farms, and in the far-field. The first batch of data are to be reported on shortly.

In addition to the monitoring data from the Wilson's Bay development NIWA has recently undertaken a broad-scale habitat survey of the seafloor of the Firth of Thames – in order to better describe the species composition, spatial distribution, and absolute abundances of the macro-benthos (M. Morrison *pers. comm.* and NIWA **unpublished data,** Dept. of Conservation funding). These data have not yet been analysed, however

preliminary indications are that *Perna canaliculus* is not abundant in the benthic samples, not as targets in side scan imagery.

There is evidence that the benthic fauna of the Firth may be subject to interannual variability, perhaps related to the ENSO-driven effects described in section 2.3.1, above. In 1999 a syndrome causing discolouration of Coromandel shellfish (oysters and mussels) occurred, the exact cause of which is as yet unknown (Diggles et al. 2000). We note that though chlorophyll concentrations were low (Figure 11, Figure 15) they were probably not sufficiently low to cause severe stress to mussels (Hawkins et al. 1999).

Greenway (1969) presents the results of extensive trawl surveys conducted throughout the Firth of Thames between 1961 and 1965 on a 1 mile x 1 mile grid. He concludes that *Perna canaliculus* was widely distributed around the western and central Firth of Thames in waters of between 3 and 10 fathoms (~5-20 m). A smaller area of mussel beds also existed around Coromandel township.   Greenway (1969) also notes that commercial catches of mussel fell from an all time high in 1961 to almost zero by 1966 – a decline which he attributed to overfishing.   *Perna canaliculus* appear to have been most abundant around Matingarahi Point (two – five times more abundant than in the majority of the remainder of the area in which Greenway indicates them to have been present). In 1961, the maximum number of live mussels collected during a 4 minute (nominally, 0.4 mile) tow (3 foot wide dredge and 1.5 inch mesh) was 271. This implies a mussel density of ~ 0.5 m^{-2}; however Greenway notes that the capture efficiency of the dredge was probably between 33% and 50%, and that commercial dredges (which had a 7.5 – 9 foot mouth and a 4 inch mesh) caught approximately ten times more mussels per minute of dredging time. Thus, it is likely that, over scales of ~ 500 m^2, the true maximum density of (relatively large) *Perna canaliculus* was circa 1.5 - 3 m^{-2} in 1961. The decline in landings (despite similar fishing effort) over subsequent years implies that densities also fell. Furthermore, given that there had been a documented commercial harvest of these mussels since at least 1927, it is possible that the estimated maximum density of 1.5- 3 m^{-2} in 1961 is less than would have been present in the absence of any historical fishing. We know of no published estimates of natural densities of *Perna canaliculus* elsewhere in New Zealand, but around Tasman Bay/Marlborough Sounds, informal estimates of density range from ~ 1m^{-2} on intertidal reefs to ~50 m^{-2} in small clusters on wharf-piles and subtidal rocks and reefs (K. Grange, NIWA, *pers. comm.*). Higher densities have been noted elsewhere: M. Morrison (*pers. comm.* & NIWA, unpublished data) reports *Perna canaliculus* densities of up to 744 m^{-2} on the channel floor in Ohiwa Harbour (though these were small mussels of 50-75 mm). Similarly, he reports that small, high density patches (~ 800 m^{-2}) of mussels occur on subtidal reefs at the south of Ninety Mile Beach. Closer to the Firth of Thames, Morrison reports densities 50 – 100 m^{-2} in Opito Bay (Coromandel) and around Waiheke Island. These latter figures are similar to **average** mussel densities on the sea-floor below mussel farms (~70 m^{-2}, maximum ~400 m^{-2}; Cole & Grange (1996)).

2.5 Faecal contaminants

Shellfish filter large volumes of water and have the potential to accumulate high levels of faecal pathogens within their tissues. Within waters to be used for shellfish growing, present guidelines dictate that the most probable number (MPN) faecal coliform median level shall not exceed 14 (100 mL)$^{-1}$, and no more than 10% of the samples shall ordinarily exceed 43 (100 mL)$^{-1}$ (Anon 1992; Brock et al. 1985). Landuse around the Firth of Thames is very much more intensive than that around the Marlborough Sounds – a correlate of this is that the Firth receives substantial inputs of faecal contaminants. These include both enteric bacteria and viruses, but monitoring is usually restricted only to counts of the bacterium *E. coli*.

Monitoring data (Environment Waikato) for swimming beaches within the eastern Firth of Thames indicates that median faecal counts are circa 10 (100 mL)$^{-1}$, however the counts are very skewed. The average is nearer 100 (100 mL)$^{-1}$ and maximum values exceed 1000 (100 mL)$^{-1}$. Comparison of these values with the microbiological guidelines suggests that **at the sites monitored** median faecal coliform counts are less than, but close to the prescribed MPN value. Since the Environment Waikato data stem from bathing beaches, it is possible that they are atypical of the Firth as a whole (as a result of septic tank leakage from nearby homes and public toilets etc.).   Thus, it is possible that levels elsewhere in the Firth of Thames are lower – particularly in view of the comparatively short life-expectancy of faecal pathogens in well-lit sea-water (Davies-Colley et al. 1994; Sinton et al. 1994; Sinton et al. in press). Equally however, faecal coliform levels within some of the rivers draining into the Firth invariably exceed the 14 (100 mL)$^{-1}$ limit. This implies that water-quality issues may preclude shellfish farming in regions which are strongly influenced by riverine inputs. We note, however, that the Waihou and Piako rivers (the two largest rivers flowing into the Firth, and also the two which are most affected by live-stock runoff) flow into the southern end of the Firth. This part of the Firth is too shallow and to permit extensive development of long-line mussel farms.

Faecal coliform concentrations are highly correlated with preceding rainfall levels. This reflects wash out of live-stock derived faecal material and 'spillage' from septic tanks and sewage treatment plants. Since it is easier to monitor rainfall than it is to monitor faecal coliform concentrations, mussel in the Marlborough Sounds farmers prefer to regulate their harvesting in relation to the preceding rainfall (Brock et al. 1985).   In particular, a three day withholding period follows rainfall in excess of 30 mm 24^{-1} hour, a four day withholding period follows rainfall in excess of 40 mm 24^{-1} hour and a five day withholding period follows rainfall in excess of 50 mm 24^{-1} hour.   A compilation of rainfall records taken at stations around the Firth of Thames indicates that there are an average of: 13, 6, and 4 days per year on which rainfall exceeds 25 mm 24^{-1} hour (*cf* the 30 mm 24^{-1} hour criterion), 40 mm 24^{-1} hour and 50 mm 24^{-1} hour respectively (data from the NIWA climate database). If we assume that the intervals between rainfall events are

such that there will be no overlap between withholding periods, then harvesting will be impossible on ~80 d y^{-1}. This is very similar to the number of days lost as a result of rainfall in the Marlborough Sounds (Brock et al. 1985).

3 Issues Specific to the Firth of Thames

The preceding review summarises our current knowledge of the Firth of Thames. Based upon this, we suggest that the following issues are of particular importance when addressing the questions of production and ecosystem carrying capacity within the Firth of Thames:

1. ENSO-related variations in the composition, biomass and community structure of the planktonic community are likely to drive inter-annual variations in shellfish productivity. This may influence the economic viability of shellfish farming in these waters. Conversely, monitoring programs aimed at detecting ecosystem effects attributable to shellfish aquaculture will need to take account of this natural variability.

2. Whilst the majority of nutrients entering the greater Hauraki Gulf stem from the ocean, approximately 30% stem from riverine inputs. These latter inputs are likely to be more important towards the south of the Firth. Given that primary production within the Firth is strongly influenced by variations in the nutrient supply rate, land-management changes may have implications for shellfish production.

3. Unacceptably high concentrations of faecal coliforms may preclude shellfish harvesting in some parts of the Firth of Thames– notably those which are strongly influenced by riverine inputs.

4. The Hauraki Gulf and Firth of Thames are home to New Zealand's largest snapper fishery, and the Firth of Thames is the key spawning ground. If shellfish farming were to reduce the abundance of microzooplankton, this may adversely influence the survival of larval snapper (and other species). Similarly, if shellfish are able to filter snapper eggs out of the water-column, this would reduce the number of fish recruiting to the population.

4 Approaches to Assessessing Carrying Capacity & Sustainability

Assessing sustainability is difficult: not least because the terms sustainability and carrying capacity are open to several interpretations. In the case of shellfish aquaculture Inglis *et al.* (2000) argue that there are four important interpretations:

a. **Physical carrying capacity** relates to restrictions placed upon the size and number of farms by factors such as geography, planning restrictions, and infrastructure.

b. **Social carrying capacity** relates to limits arising from impacts which are deemed to be socially unacceptable. These might include reduced visual amenity, and access restrictions.

c. **Production carrying capacity** is defined as the stocking density which allows the sustainable harvest to be maximised.

d. **Ecological carrying capacity** is defined as the quantity of farm development beyond which ecological impacts are deemed to become unacceptable.

Inevitably, different parties attach differing levels of importance to each interpretation. The task of arbitrating between parties, and ultimately, making a (largely subjective) decision as to which interpretation should carry the greatest weight for the area at issue falls upon central government and regional authorities. We suggest that this process can be conducted most efficiently and transparently by explicitly defining thresholds appropriate to each interpretation (we will refer to such thresholds as **issue** thresholds). It is to be hoped that the **issue** thresholds will be defined primarily by objective criteria. Subsequently, a decision must be made as to which interpretations of sustainability should carry the most weight for the area under consideration. This judgement may entail subjective rather than objective reasoning.   In turn, the judgement will influence which of the candidate, issue thresholds are eventually adopted as the final list of criteria by which an individual application should be assessed (and, if approved, against which the farm's impacts should be assessed).

When determining issue thresholds, three distinct exercises are required:

i.   Definition of characteristics by which deviations from sustainability are to be measured, together with the associated sampling protocols etc.

ii. Definition of threshold, or trigger-levels of deviation at which some form of management intervention will be required

iii. Definition of the statistical tests by which to judge whether the characteristic in question has surpassed any of its associated threshold(s).

We will address these three issues within the remainder of this section of the report. At the very end of this section, we will introduce a table (Table 5) of candidate characteristics that we believe should be used to determine issue thresholds (for production and ecological capacity).

4.1   Definition of characteristics

When seeking to define the characteristics by which to monitor impacts one is faced with the problem that public concern is concentrated upon the so-called 'charismatic megafauna' (birds, mammals, some fish etc.), whilst a mussel farm's most direct impacts are likely to be upon a very different group of organisms: those which the mussels consume or compete with.

The charismatic megafauna may be adversely influenced by direct impacts of mussel farming (loss of habitat through physical exclusion or disturbance effects etc.), or indirectly through changes in the local abundance of their prey. The direct impacts of mussel farming upon megafauna have received very little attention. One opinion holds that these fauna will quickly become oblivious to disturbance by mussel barges, and may even benefit from the increased habitat diversity afforded by mussel farms and from the opportunistic fauna associated with farms. Others argue that the fauna will flee from the disturbance. It is possible that some species will adapt whilst others will flee! In an attempt to resolve the issue for one species, MfE have recently requested tenders for research into the impacts of mussel farming upon King Shag in the Marlborough Sounds.

The charismatic megafauna may also be affected by changes in the abundance of their prey. Unfortunately, there are often two or more trophic levels between the mega-fauna and the organisms that shellfish consume or compete with. There are very few cases where ecologists have successfully predicted how changes at the bases of a food-web will influence populations near its top – particularly when the changes at the base may be small in magnitude, or spatial extent (Ross, A.H. 2002). For this reason, it is argued that the best practicable approach is to: (a) minimise impacts at the base of the foodweb (hoping that this will be sufficient to protect organisms higher in the foodweb); and (b) ensure that there is adequate monitoring of the higher organisms in order that any

changes are quickly detected (as a 'backstop' against the possibility that the mega-fauna are influenced by subtle changes lower in the food-web, or by other (non-food-related) habitat changes induced by the farming activities.

There is no universally agreed set of characteristics by which to judge the impacts of aquaculture. Nor are there any agreed standards by which to determine the associated thresholds. Rather, characteristics and thresholds are usually determined on a case by case basis. To some extent, the need to take site-specific features into account makes this inevitable. Equally, there are numerous fundamental characteristics that all shellfish farms will exhibit. This suggests that it should be possible to define a corresponding 'core-set' of characteristics which should be measured in all cases. Definition of such a set would have obvious practical benefits. They would facilitate *a-priori* budgeting of farm operations, streamline resource consent proceedings and promote better definition of data-handling and data-reporting standards with consequent efficiency gains.

The definition of site-specific characteristics requires negotiation between relevant parties. We believe that this is also the case for the 'core-set' characteristics. Thus, within this document, we do not propose to develop a definitive set of core characteristics. Rather, we offer a list of candidate characteristics (Table 5) together with a justification of each and suggestions regarding data and issues which might provide means by which ecologically relevant thresholds might be defined for each. We emphasize that this list is offered as a means of initiating discussion with all interested parties. We acknowledge that the list is not an exhaustive compilation of possible candidates, but we hope that it is a sufficient one (i.e. that there are no other characteristics which should also be included within the list). The list should certainly not be regarded as a definitive list of necessary characteristics. We acknowledge that some parties may argue that one or more of the characteristics are not required, whilst others may argue that our list is inadequate.

We follow Hatton *et al.* (2002) and distinguish between characteristics based upon (a) specific biological variables, (b) surrogate biological variables, (c) multivariate variables, and (d) abiological variables (an extension of Hatton's 'Sediment Characteristics'). We will not reiterate the variables listed in Tables 3-6 of (Hatton et al. 2002) – which lists a number of characteristics which might be used to measure farm effects upon the benthos. We will, instead concentrate upon variables which might be used as measures of impacts within the water-column. In general, we suggest that the characteristics listed in Table 5 should be measured along one, or more radial transects extending from the centre of the farm to a distance substantially beyond (several hundred meters to km) the farm's perimeter. The total number of radial transects, the number of sampling sites per transect and location of each such site should be determined on a site-specific basis taking account of prevailing currents and site-specific issues (e.g. nearby habitats deemed to be particularly valuable or sensitive).

4.2   Definition of Thresholds

Perhaps the most difficult part of any assessment of sustainability is that associated with defining objective **issue thresholds** (*c.f.* characteristics by which to measure impacts) at which operations are judged to become unsustainable. With respect to production and ecological capacity, we believe that there are three approaches which can help in defining thresholds:

i.   reference to historical data

ii.   application of simple (linear, equilibrium) models

iii.   application of more complex, dynamic models

4.2.1   Historical data

We distinguish between three types of historical data: those specific to the region in question (e.g. time-series of nutrient and phytoplankton abundance etc.), those concerning analogous ecosystems elsewhere and, finally, those providing detailed information concerning the environmental requirements for (some of) the species (or analogous) species characteristic of the region in question.

Historical data provide evidence of how the ecosystem has behaved in the past. Historical data provide one form of 'control' data against which to test for change following aquaculture developments (i.e. during the monitoring phase). Historical data are equally important during the assessment phase. They make an assessment of the 'natural' characteristics (mean and variability) of the system possible. Comparison of the historical range of values associated with each characteristic with the values which are predicted to pertain following aquaculture development provide an indication of the degree of stress which the system is likely to be put under.

In situations where there are no historical data, it may be possible to use data from analogous regions in a manner akin to those in which historical, region-specific data are used. The analogous system may have no aquaculture industry – in which case it provides an indication of the state which the present region (e.g. Firth of Thames) may currently be in. Alternatively, the analogous system may have an aquaculture industry of its own – in which case it provides an indication of the state to which the present region (e.g. Firth of Thames) may evolve. If one is very lucky, there are before- and after- data for the analogous system – in which case it is possible to make an assessment of how much change is likely to occur within the present region.

The third type of historical data concerns the environmental requirements of individual members of a region's ecosystem. In the case of aquaculture within the Firth of Thames, such data might include: relationships between individual growth (or adult reproduction) and phytoplankton abundance (for mussels and other grazers of

phytoplankton). Other relevant data might include: the extent to which the eventual size of the adult population of (for example) snapper is determined by survival rates during the egg and larval stages. In most cases, data of this type are most easily gathered in laboratory studies (subject to caveats regarding laboratory artefacts).

4.2.2 Simple Models

Simple mathematical models provide a means by which to make qualitative predictions of the magnitude of change that may be associated with aquaculture development. In essence, models provide a means of extrapolating into the future based upon present knowledge. Clearly, the quality of the prediction depends not only upon the quality of present knowledge, but also upon the quality of the extrapolation procedure.

As the term suggests, simple models depend upon data which is comparatively easily gathered and adopt unsophisticated methods of extrapolation. In the latter respect, they often make an assumption that the system is in equilibrium (i.e. they ignore day-to-day, seasonal, and inter-annual variability, and aim to predict the long-term average state to which the modified system may evolve; they make no pretence to predict how quickly the modified system will evolve to this new state). Furthermore, simple models are usually linear (in the context of feeding, for example, this implies that the consumption rate is linearly dependent upon the prey concentration). In reality, many biological processes are non-linear. For example, consumption rates are usually saturating functions of food-density and growth rates may depend as much upon the 'quality' of the food (species composition) as much as it does upon the total quantity of food. It is possible to use a linear model to predict the value of a (truly) non-linear process with moderate accuracy **provided** that one does not endeavour to extrapolate 'too far' from the known (current) conditions.

The great merits of simple models are that: (a) the data which they require is often already available (or can be gathered at only moderate expense in terms of both time and money), and (b) the models themselves are comparatively easy to understand, construct and apply.

We describe several simple models within sections 5-7 of this report. These include: the model used to provide depletion estimates (section 6), the ECOPATH trophic model (section 7.2), and the inert-tracer model (section 5.3.2.1; which is sufficiently complex that it may deserve to be classified as a "Complex model").

4.2.3 Complex Models

These seek to remedy the deficiencies of simple models. They usually adopt more appropriate (non-linear) descriptions of processes. They may consider more interactions than those included within simple models, and they often explicitly consider issues of temporal change. In theory (and sometimes, in practice), complex models can be used

to predict the consequences of more extreme change (because the extrapolation is based upon more realistic assumptions), and they can provide more detailed information (because the models usually incorporate more variables, and explicitly address issues related to space and time).   Inevitably, the disadvantages associated with complex models are: they demand more sophisticated data (less readily available, more costly to gather), they are more difficult to formulate and implement, and whilst they can yield important and sophisticated predictions, it can take considerable effort to fully understand the processes governing a particular outcome.

We describe several complex models within sections 7.2 of this report. These include: the production-capacity model, the biophysical model and the lattice-gas-automata model.

4.3   Definition of Statistical Tests

Within the fields of environmental sciences and management (as in many others), the usual means of determining whether two data sets differ is through the use of one-, or two-sided tests of a point null hypothesis. McBride & Ellis (2001) makes a persuasive argument that this is inappropriate. The fundamental reason is that we know the null hypothesis to be false *a-priori*. McBride & Ellis recommend the use of tests based upon interval hypotheses. These tests adopt a null hypothesis which is not known to be false *a-priori*. Consequently, the outcome of the test is less dependent upon the sample-size (i.e. more robust), and also more informative. The reader is referred to Appendix 1 for further details. Here, we merely note that interval tests are widely used in medical applications such as trials of pharmaceuticals and recommend that they be adopted as a means of determining whether a farm's effects have exceeded predefined thresholds (the trigger-levels of Hatton *et al.* (2002), and Table 5 of this report).   The predefined thresholds may be set by reference to judgments regarding the magnitude of effect which is considered to be ecologically significant (*cf* statistically significant). Thus, interval tests provide an explicit means of determining whether or not the impact of an individual farm, or the cumulative effects of many farms surpass those deemed to be ecologically significant.

Table 5

Candidate characteristics by which deviations from the pelagial-phase
ecological carrying capacity of the Firth of Thames might be assessed.

Characteristic	Core or site-specific?	Sampling Frequency	Rationale	Data which might be used to define trigger levels
Nutrient Concentrations	Core	Weekly to monthly	Nutrients (esp. nitrogen) frequently limit primary production in the Firth of Thames	(a) Departures from 'control' site concentrations. (b) excursions beyond historical (pre-farming) ranges characteristic of prevailing ENSO conditions. (c) changes in phase of seasonal nutrient cycle
Size fractioned phytoplankton concentrations	Core	Weekly to monthly	Indicator of food availability to other organisms feeding upon phytoplankton	(a) Chlorophyll concentrations in 'control' areas. (b) half-saturation coefficients for feeding / growth in key grazer taxa
In-situ chlorophyll fluorescence	Core	Weekly to monthly	Large areas can be surveyed quickly thereby compensating for the high spatial variability which is common in phytoplankton distributions	
Abundance of key zooplankton taxa (microzooplankto o, copepoda, eggs & larvae of benthic organisms)	Core	Monthly	Indicator of changes in abundance / community composition of phytoplankton & microzooplankton. Indicator of food availability to higher trophic levels (fish etc)	(a) Abundance in 'control areas' (b) Half-saturation coefficients for feeding / growth in key planktivore taxa
Survival of larval fish	Site-specific	Several times during spawning season	Variations in larval survival in fish usually determine adult stock-size	Survival relative to expected survival based upon historical data (e.g. SST-relationship for snapper (Francis 1993))
Abundance of selected post-pelagial stages of benthic organisms (incl. tidal zone organisms)	Site-specific	Monthly to annually	Indicator that farming is reducing survival/recruitment success of the egg/larval stages of these organisms, or reducing productivity of adult stages (reduced phytoplankton availability etc).	(a) Departures from control site abundances (b) Departures from historical norms
Population size & distribution of charismatic megafauna	Site-specific	annual	Indicator that farms are having adverse trophic or disturbance effects	(a) Departures from control site abundances (b) Departures from historical norms

5 Assessment of Sustainability Based upon Present Knowledge

5.1 Physical Carrying Capacity

Depth is likely to be the principal determinant of physical carrying capacity within the Hauraki Gulf. Mussel farms are rarely placed in water less than 10 m of depth. Thus, much of the southern and central Firth of Thames is too shallow to sustain large-scale mussel aquaculture (Figure 1) – though oyster farming may be possible.

To date, the majority of mussel farm developments in New Zealand have been restricted to areas of muddy sea-floor. This has been based upon a perception that such areas have lower amenity value, and support comparatively sparse, but resilient flora and fauna.   We know of no published studies which could be used to determine whether muddy-floor areas are more resistant to mussel farm developments than are sandy- or rocky shore ones – though (Cummings et al. 2001) report that, downstream of Horse mussel (*Atrina zelandica*) beds, there were larger changes in the benthic communities of sandy shore communities than there were in muddy shore communities.

Within the Firth of Thames, restriction of farm developments to muddy shore sites would be unlikely to influence the majority of pending applications, but would preclude developments in the northern-most parts of the Firth – which are sandy (Figure 2). Phytoplankton concentrations are known to be high in this area (Figure 13). Thus, it is possible that operators will seek to develop farms in this area in the future.

Based upon surveys of snapper eggs and larvae within the greater Hauraki Gulf (*cf,* Firth of Thames alone), it was concluded that the majority of spawning takes place in waters of less than 30 m depth. If farm-effects were considered likely to adversely impact upon the survival of snapper eggs and larvae, some parties might advocate excluding aquaculture from areas in which the water-depth is less than 30 m. This would exclude aquaculture from most of the Firth of Thames! Clearly, before adopting such a policy one would need to consider whether there is a need to determine how important the Firth of Thames (*cf* greater Hauraki Gulf) is for snapper eggs and larvae, and perhaps also the extent to which there Firth of Thames and Hauraki Gulf fish intermingle.

5.2   Production Carrying Capacity

Average algal concentrations in the northern part of the Firth of Thames are circa 1- 2 µg Chl*a* L^{-1} (Figure 12, Figure 13, Figure 15, Table 3). This is higher than is typical within the Marlborough Sounds (Table 3). Hawkins et al. (1999) concludes that the assimilation rate of *Perna canaliculus* saturates (reaches a maximum) at a chlorophyll concentration of circa 2 µg Chl*a* L^{-1} . Thus, chlorophyll concentrations in the northern Firth of Thames should enable mussel growth rates which are greater than those of in the Marlborough Sounds.

Simulation models developed for the Marlborough Sounds demonstrate that both overall meso-scale (10s – 100s of km^2) farm density, and the distribution of farms within the meso-scale can have substantial impacts upon mussel production (see section 7.2 for an example from the Marlborough Sounds). At present, we have no equivalent model for the Firth of Thames. In principal, however the Marlborough Sounds model could be adapted for application within the Firth of Thames.

5.3   Ecological Carrying Capacity

5.3.1 Overview

As we indicated earlier, there are no definitive means by which to assess the ecological carrying capacity (or deviations there-from) of an area. Within Section 4 of this report we listed several candidate characteristics which might be used to estimate deviations from the ecological carrying capacity. Within this sub-section we will make some semi-quantitative estimates of how some of these characteristics may be influenced by shellfish farms.

Most of the characteristics poposed in Section 4 are based upon living organisms. We draw a distinction between three types of organism:

i.   primary organisms (shellfish 'prey'): those which are filtered out of the water column by the shellfish (whether subsequently consumed or ejected within pseudo-faeces),

ii.   secondary organisms (mussel 'competitors'): those which suffer as a result of depletion of the primary organisms (predominantly, other planktivores).

iii. Tertiary organisms: those which are affected for other reasons (eg habitat change on the sea-floor or immigration of new predators onto the farm).

In what follows, we will consider only primary and secondary organisms. Inglis *et al.* (2000) list several organisms which fall into the tertiary category (spotty, *Notolabrus celebrus*; predatory starfish, urchins etc. tend to accrue under mussel farms, whilst the

abundance of burrowing fauna tend to fall), but to date we can make few quantitative statements regarding such organisms.

It is important to appreciate that, though many individuals (from among the populations of primary organisms) may be removed from the water that passes through a shellfish farm, others are likely survive their passage through the farm. These will continue to grow and reproduce, and in time, the population may return to a state which cannot be distinguished from that which would pertain had there been no depletion. In addition, mixing of impacted and pristine (defined in section 1.2) waters will introduce more individuals into the depleted water, and these 'colonists' will not only 'dilute' the original impact by their mere presence, but also contribute to the biological recovery process through growth and reproduction. The relative importance of 'survivors' and 'colonists' to the recovery process depends upon two factors: the generation time (inverse of the growth rate) of the organism in question and the rate of mixing between impacted and pristine water. Where the growth rate exceeds the mixing rate, recovery will be dominated by *in-situ* re-growth of the surviving population. Where mixing dominates, recovery will be dominated by 'dilution' of the impact signal as the impacted waters become dispersed within the pristine waters.

Numerous factors influence the processes governing *in-situ* biological recovery, but shellfish farms are likely to influence only two: the quantity (and quality) of food, and the numbers of competitors and predators/pathogens. The factors which govern the rate of mixing between pristine and impacted waters are, perhaps less clear. The most obvious is the degree of turbulence within the water-column. This determines the area-specific rate of exchange between pristine and impacted waters. A second, less obvious determinant of the rate of mixing is the surface area of the interface between pristine and impacted waters. Little exchange is possible if the two water-masses interact over only a small surface area. Conversely mixing will be more rapid when the two water-masses interact over a large surface area.

5.3.2 Comparison of natural densities of *Perna canaliculus* with those in marine farms

In section 2.4.3 we noted that survey data indicated the *Perna canaliculus* used to be found in on the bed of the central and western Firth of Thames in waters between 5.0 and 20 m. Based upon 1961 data, Maximum densities were probably ~ 3 m^{-2} (measured over ~ 500 m^2), but this density was attained only in the region immediately around Matingarahi Point. In general, densities declined in subsequent years of Greenway's study (attributed to overfishing), but it not possible to determine whether the figure of 3 m^{-2} is representative of maximum attainable densities or whether (supposed) overfishing had already depleted the beds (landings peaked in 1961, but had been almost as high over the preceding four years). Informal observations suggest that densities can attain ~50 m^{-2} on sub-tidal reefs and rocks elsewhere in New Zealand (K. Grange, NIWA, *pers. comm.*).

The abundance of mussels declined dramatically during the early – mid 1960s. Anecdotal evidence suggests that they have not recovered since then. This raises the possibility that the addition of mussels (through marine farming) merely reintroduces mussels into an area in which they were once wide-spread. This argument ignores complications arising from the fact that farmed mussels live, and feed in mid-waters, whilst the earlier, natural populations lived on the sea-bed. Nonetheless, it is worth asking the question: how do mussel densities within marine farms compare with historical densities within the Firth of Thames?

To address this question, it is important to ensure that we calculate (farmed) mussel densities at a scale which can be legitimately considered similar to that at which Greenway made his measurements. Thus, we suggest that estimates of mussel density based upon an individual dropper line would be inappropriate. Rather, we choose to determine the mussel density at the scale of an individual farm (ie number of mussels within a farm divided by the area of sea-bed enclosed within the farm's boundaries.

The details of each individual farm will differ, but we take those of the Wilson's Bay farm as an example. The consent conditions for this operation dictate that there shall be no more than 2 mussel lines (backbones) ha^{-1}. This implies a maximum of 4 dropper-lines per ha. Backbones are 150 m long and support 3600 m of dropper line (2400 m (100 m)$^{-1}$). The mussel density (number per m of dropper line) is dependent upon the age of the mussels, but we adopt a figure of 140 m^{-1} as being appropriate given the mesh size used by Greenway. At present, farms consist of two parallel backbones 100 m apart. This implies that a farm occupying an area of 1 ha will contain ~6.72×10^5 mussels (2 backbones separated by 100 m, each of 150 m x 3600 m per dropper line x 140 mussels per m divided by an area of 100 m (line-spacing) x 150 m (backbone length)). Thus, the farm-scale density of mussels is ~ 67 mussels m^{-2}. The resource consent further stipulates that parallel backbones should be no less than 22 m apart.   If farms were configured in this latter manner, the mussel density **within** the area enclosed by the lines (*cf* the enclosing 'buffer-strip' area which would be required to meet the stricture that the average backbone density should not exceed 2 ha^{-1}) will be ~300 m^{-2} [(2 backbones x 150 m x 3600 m per dropper line x 140 mussels)/(150 x 22)].

These calculations imply that, when operating to the full extent permissible within the consent, mussel densities within the Wilsons Bay farm will be at least an order of magnitude greater than the maximum density recorded anywhere within the Firth of Thames in 1961. Indeed, they will be of the same order of magnitude as the maximum densities which *Perna canaliculus* has been reported to attain around the New Zealand coast (section 2.4.3).

5.3.3 Farm Footprints

In this part of the report we derive some preliminary estimates of the radius to which a farm's influence may extend. The derivation has two components: firstly, we simulate the current-driven transport of many thousands of biologically inert particles away from a nominal farm-location. This provides a good indication of how fast a farm's impacts may be transported around the Firth of Thames, and also, some indication of the degree to which they become diluted in the meantime. In the second part of the derivation, we make a comparison of these transport speeds with the likely rates of *in-situ* recovery (*c.f.* recovery through dilution/colonisation effects). This enables us to make a **qualitative** assessment of the magnitude of any far-field farm effects for materials / organisms having differing intrinsic regeneration rates. We address the first and second part of the derivation under the sub-sections 5.3.2.1 and 5.3.2.2 .

The results presented within this report are based upon tracer releases made from six sites. These release sites do not represent proposed marine farm sites, but are chosen to represent the different geographic areas and bathymetries of the Firth of Thames. Within the Firth of Thames, there is a strong relationship between the tidal-residual current speed and water-depth. Thus, the results of the dispersal study are not strongly influenced by small changes in release site location, and provide a good general tracer footprint indication. The Firth was partitioned by eye into two 'inner-Firth', two 'mid-Firth' and two 'outer-Firth', or 'Gulf' areas (Figure 1). Each area is of approximately equal surface areas, but have differing mean depths (sites 1-6: 1, 3, 10, 10, 25 and 35 m respectively). Tracer particles were released from the mid-point of each area.

The footprint analysis was made for two tracer types: neutrally buoyant and negatively buoyant (henceforth: 'settling'). Over one 12.5 hour tidal cycle, tracer particles were continuously released into the water column at a rate of 300 per hour. Each particle was released at a random depth between the surface and the lesser of: sea-floor depth, or 15 m water depth (the expected maximum depth of a marine farm). After its release, each particle were allowed to disperse (and, for sinking particles, settle) in accord with the tidal- and wind-driven currents. Particles which sink to the sea-floor were assumed to move no further. A settling velocity of 0.02 m s^{-1} was employed for this study, based on the expected settling rates of faecal material from a 6 cm mussel (N. Hartstein, NIWA, *pers. comm.*).   Thus, our settling particles provide an analogue for mussel faeces. The neutrally buoyant particles provide a crude analogue for other shellfish 'products' (e.g. water depleted of phytoplankton and enriched in nutrient).

We consider 5 different wind scenarios (no wind, and winds of 15 m s^{-1} from the NW, SW, NE, SE). Given the importance of wind in determining the residual pattern of water circulation (see below), these five scenarios encompass a wide range of likely dispersal patterns (from poor dispersal under calm conditions, to unusually extensive/rapid

dispersal at 15 m s^{-1} – which is above average for the region).. Additional details of the simulations are provided in Appendix 2.

Distribution of inert tracer

As a preliminary to describing the specific distributions arising from our six release sites, we present the simulated near-surface residual circulation patterns (ie net current velocities over one tidal cycle) in the Firth of Thames for tide and wind-driven currents (Figure 19 and Figure 20; see also (Oldman & Senior 2000)). The near-surface residual currents are shown because they incorporate the strongest wind-driven effect and consequently the greatest advective dispersion.

 shows that residual tidal currents are small, but wind stress creates stronger residual circulation (Figure 20). Residual currents tend to be greater in the northern part of the Firth of Thames, but the absolute speed and direction of the surface water circulation is very strongly dependent upon the wind-direction.

Figure 21 to Figure 25 shows the migration of a neutrally-buoyant tracer uniformly released to a depth of 15 m in the Firth of Thames, under various wind scenarios, while Figure 26 to Figure 30 shows the migration of a settling-tracer. The tracer footprints are shown after 75 hours – a total of 5 tidal cycles after the initial tracer release cycle finished. This period was chosen as being close to the maximum expected duration that a wind will blow from the same direction.

The logarithmic colour scale enables very low concentrations to be easily seen (if all the tracer were contained in one cell, it would have a Log$_{10}$ concentration of ~3.6). Under calm conditions the plume movement is predominantly controlled by the ebb and flood tidal currents. Turbulent diffusion will also disperse the tracer horizontally and vertically, although at a slower rate.

Generally, the further north the release site is, the larger the footprint for the neutrally buoyant tracer (Figure 21). This occurs because the tidal residual current strength increases in the upper Firth of Thames with a resulting increase in tracer dispersion (Oldman & Senior 2000). Under calm winds (tidal currents only), settling particles are predicted to fall to the sea-floor almost directly under the release site – particularly in the south of the Firth, where current speeds are lower and the water is shallower. Thus, as characterised by the concentration of those particles which still remain in suspension, the footprint of the tracer is very small (Figure 26). This implies that the footprint of settled tracer is also small, but correspondingly intensified.

Wind stress can drive currents similar in strength to the peak tidal currents (Oldman & Senior 2000), and so they provide considerable variability in current patterns in the Firth of Thames. Due to the long and narrow shape of the Firth, both tidal and wind-driven currents tend to flow along the alongshore axis (north-south), so the winds tend to reinforce either the ebb or flood tide depending on wind direction (Oldman & Senior

2000). This contributes to current variability and promotes tracer dispersion in the north-south direction, consequently a larger north-south spread was observed in the conservative-tracer footprints when 15 m s^{-1} winds were imposed (Figure 22-Figure 25). For example, during a northeast wind (Figure 22) the plume will migrate towards the southwest into the inner Firth of Thames. The highest concentrations of tracer however are transported by the near-coast alongshore currents and remain close to the coast (Oldman & Senior 2000).

Figure 19.
Tidal residuals in the near-surface layer (0-2 m) during calm conditions (Oldman & Senior 2000). On average, the ebb and flood tides in most places are nearly equal and opposite, consequently tidal residuals are mostly small.

NIWA / Waikato Earth Sciences   Model

Mean Velocity vector k = 1

0.1 m/s
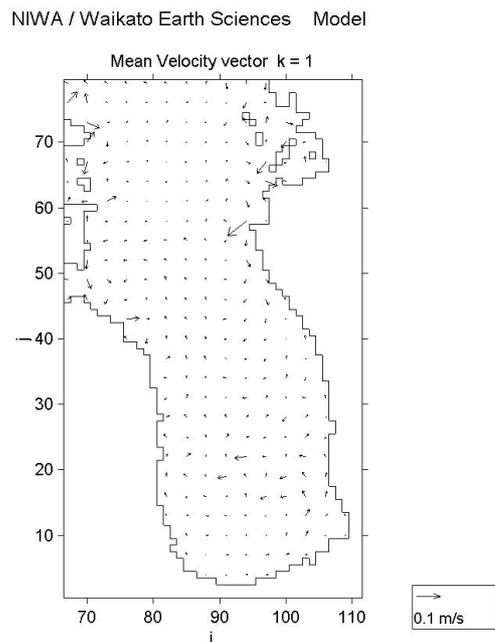
Figure 20.

Tidal and wind-driven residual currents in the near-surface layer (0-2 m) during 15 m s^{-1} winds from (top-left to bottom right) the NE, SE, SW and NW (Oldman & Senior 2000).
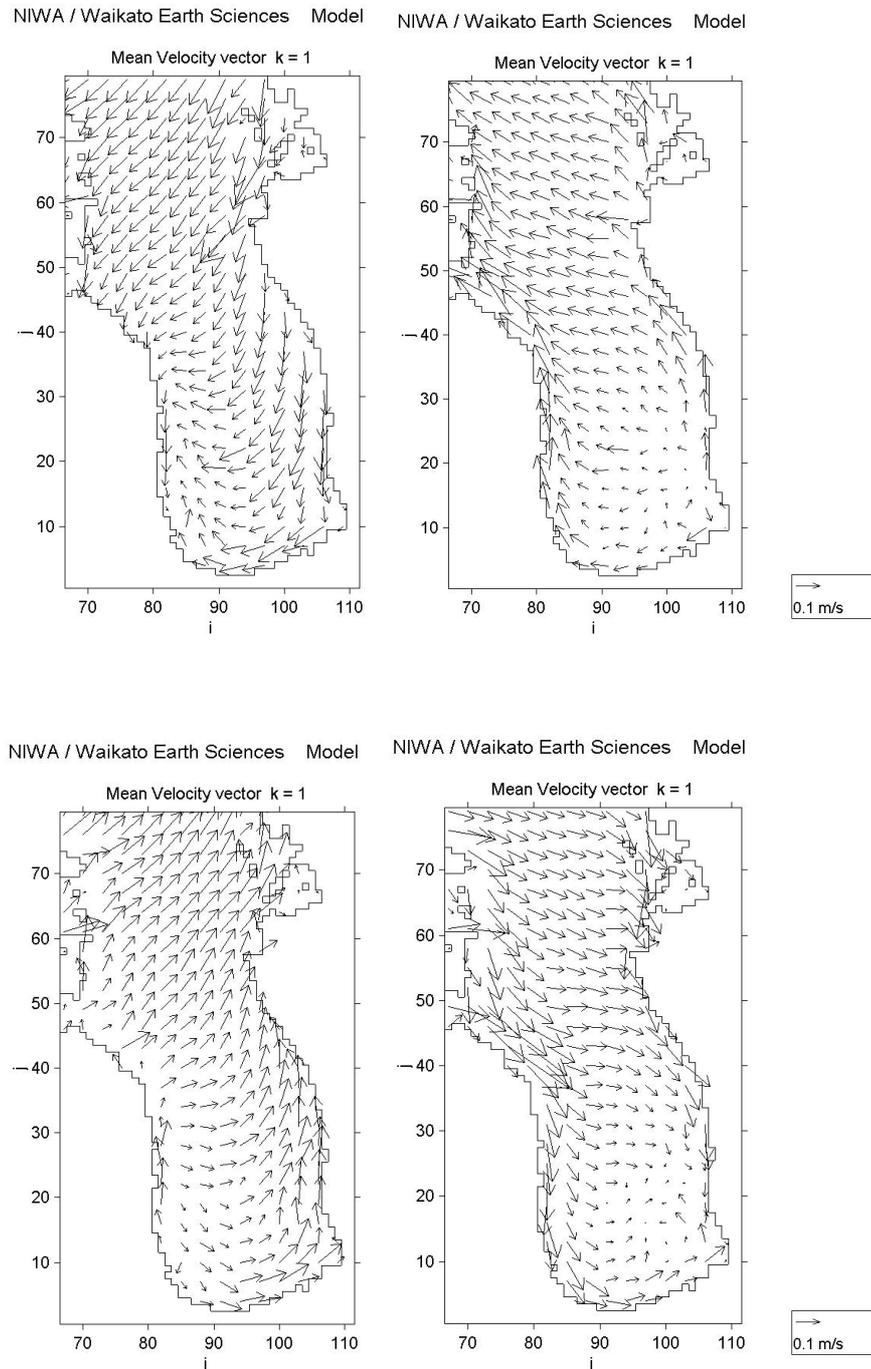
In addition to north-south spreading, winds approaching from the northwest and southeast (in particular), promoted eastward and westward migration of the conservative-tracer plume. This was particularly true of the deeper and more exposed sites 5 and 6, which exhibited plumes extending from the release sites all the way across the Firth of Thames to the shoreline. Thus footprints from marine farms in the upper Firth of Thames can be expected to spread around Waiheke Island during southeast winds.

Settling-tracer plumes exhibited similar trends to conservative-tracer plumes, but tracer continued to rapidly settle out of the water column even under exposure to 15 m s^{-1} winds, and so large changes in settling-tracer footprints were not observed (Fig. 27 – Fig. 30).

We can gain some impression of the importance of mixing processes in promoting recovery around the farm by comparing the expected number of particles which would be present in the grid-cell containing the release site at the end of the initial release period **in the absence of any transport/mixing** (i.e. 3660 particles) with the simulated (**incorporating transport and mixing**) particle concentrations **within this grid-cell at the end of the first tidal cycle** (i.e. at the end of the period of particle release). Transport will tend to disperse the particles away from the release site, thereby diminishing the impact of the farm. Thus, the highest impacts can be expected under calm conditions, and we make the comparison only for these conditions (Table 6). Comparison of the figures for non-sinking particles suggests that, even close to the farm, mixing may reduce particle concentrations by more than 80%; except in the shallow, southern Firth, deposition contributes relatively little to the removal of particles from the source area. It should, however, be remembered that whilst these figures are "worst-case" in the sense that they are for calm (= weak transport) conditions, they assume no prior contamination of the water. In areas where tidal residuals are small, or areas where the inflowing water is already impacted by other, upstream farms, mixing processes will be less effective in promoting recovery.

Table 6

Predicted particle count (grid-cell^{-1}) within the grid-cell containing the release site at the end of the release period under calm conditions. These values are compared with a value of 3660 (expected particle count within the grid-cell containing the release site in the absence of transport **and** sinking) to derive the % reduction figures.

Particle	Characteristic	Site Number					
Type		1	2	3	4	5	6
Non-sinking	Particle count	603	490	692	295	324	501
sinking	Particle count	1	107	1	115	447	646
Non-sinking	% reduction	84	87	81	92	91	86
sinking	%reduction	100	97	100	97	88	82

Figure 21.

Depth-averaged concentration (Log$_{10}$ scale) of a conservative-tracer released five tidal cycles previously at sites 1 to 6 under calm conditions into a stratified water column. Plot layout follows release site location (Figure 1). If all the tracer-particles which were to remain within the 750m x 750 m cell in which they were released, the particle concentration would have a Log$_{10}$ value of ~3.6.
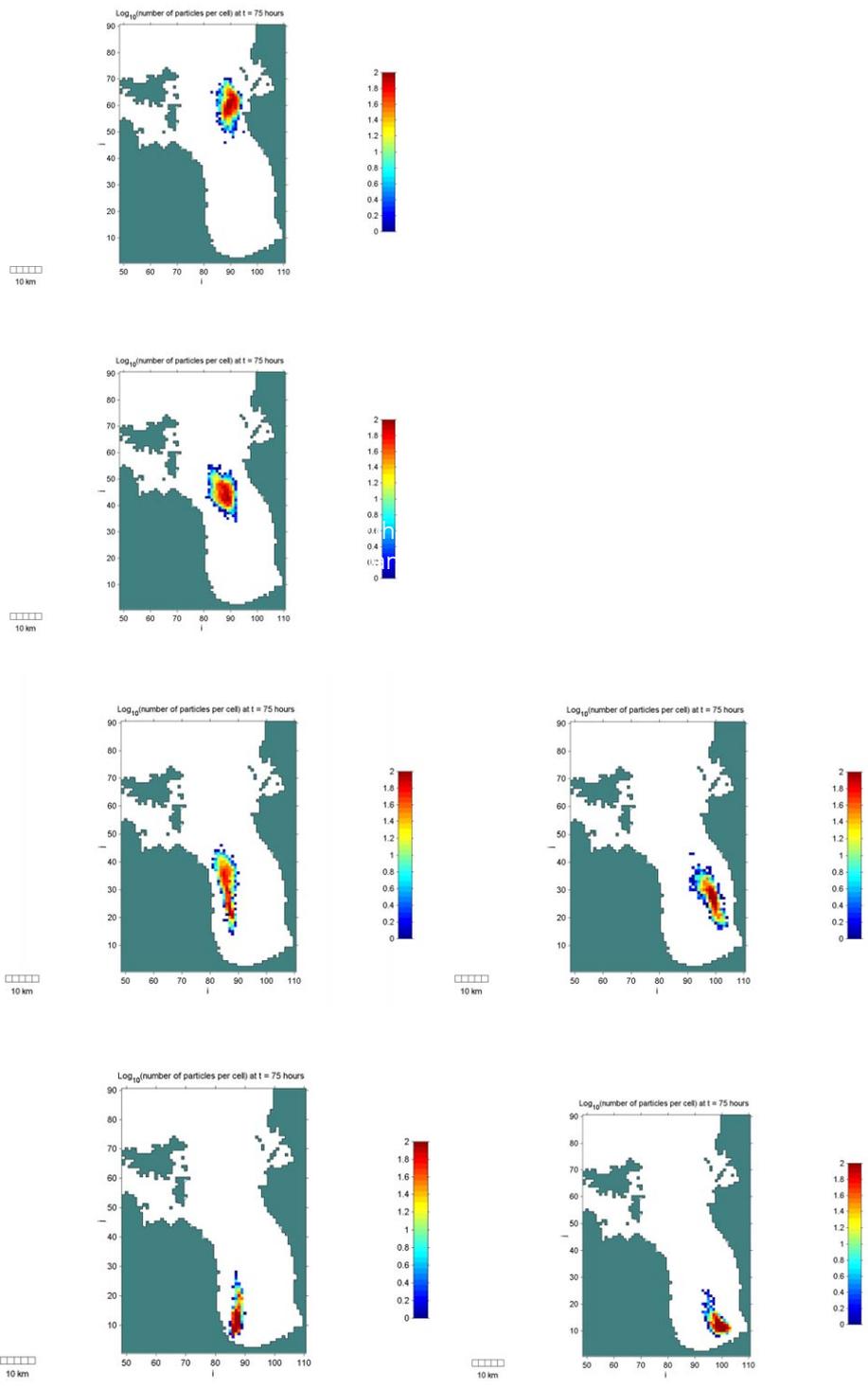
Figure 22.
Depth-averaged concentration (plotted as Log_{10}) of a conservative-tracer five tidal cycles after release at sites 1 to 6 under a 15 m s^{-1} NE wind into a stratified water column. Plot layout follows release site location (Figure 1). If all the tracer were contained in one cell, it would have a Log_{10} concentration of ~3.6.
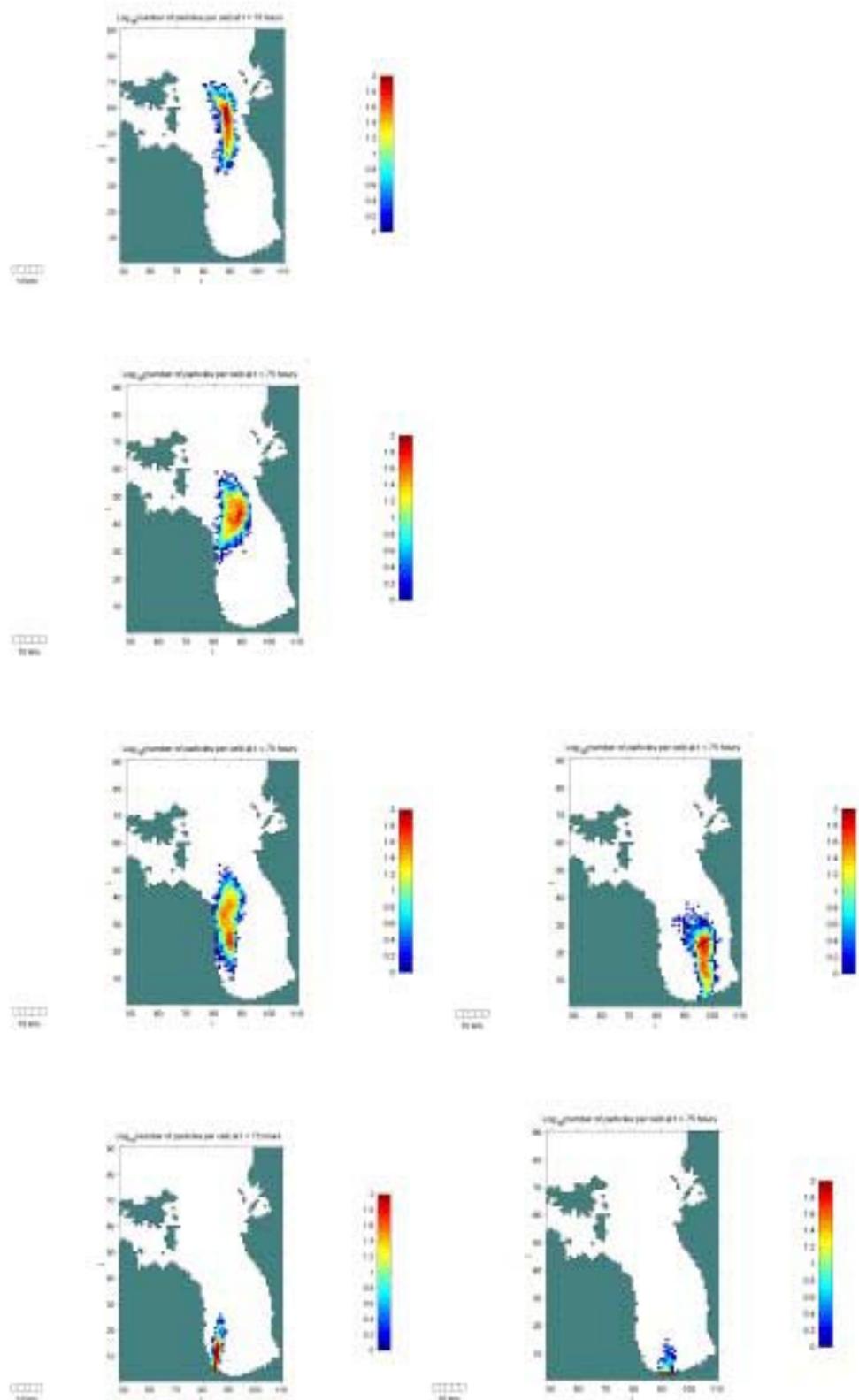
Figure 23.

Depth-averaged concentration (plotted as Log_{10}) of a conservative-tracer five tidal cycles after release at sites 1 to 6 under a 15 m s^{-1} SE wind into a stratified water column. Plot layout follows release site location (Figure 1). If all the tracer were contained in one cell, it would have a Log_{10} concentration of ~3.6.
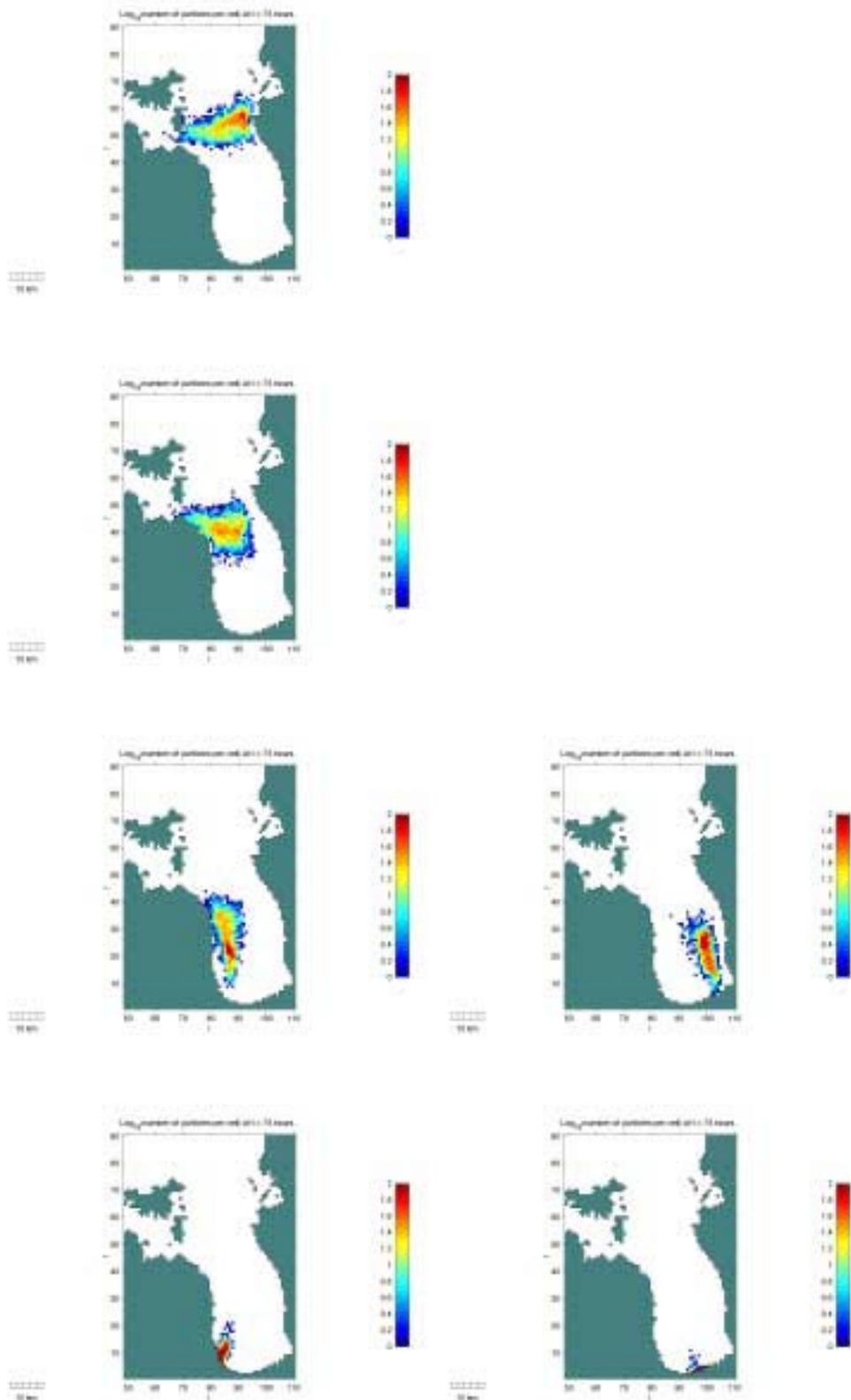
Figure 24.

Depth-averaged concentration (plotted as Log_{10}) of a conservative-tracer five tidal cycles after release at sites 1 to 6 under a 15 m s^{-1} SW wind into a stratified water column. Plot layout follows release site location (Figure 1). If all the tracer were contained in one cell, it would have a Log_{10} concentration of ~3.6.
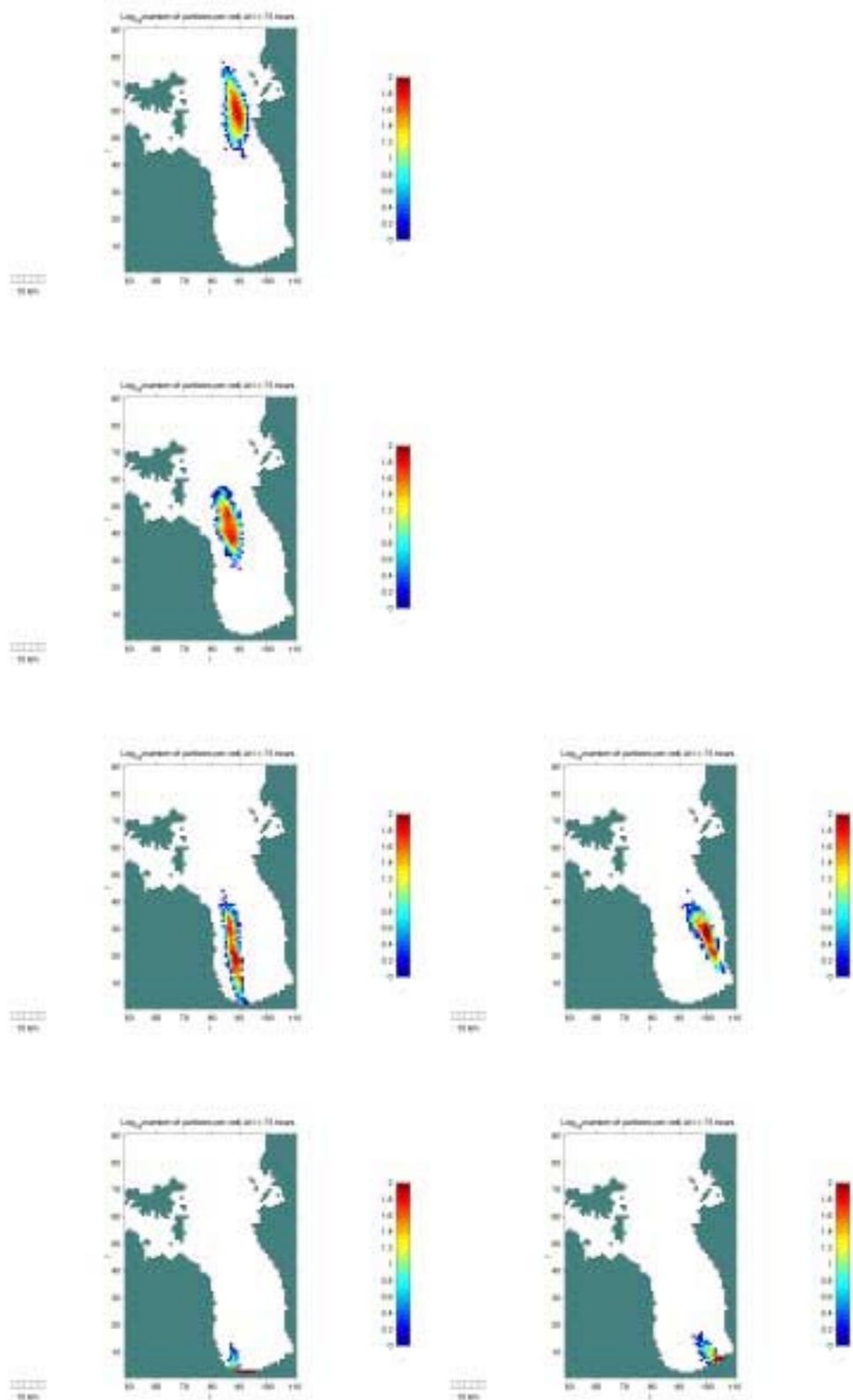
Figure 25.

Depth-averaged concentration (plotted as Log_{10}) of a conservative-tracer five tidal cycles after release at sites 1 to 6 under a 15 m s^{-1} NW wind into a stratified water column. Plot layout follows release site location (Figure 1). If all the tracer were contained in one cell. it would have a Log_{10} concentration of ~3.6.
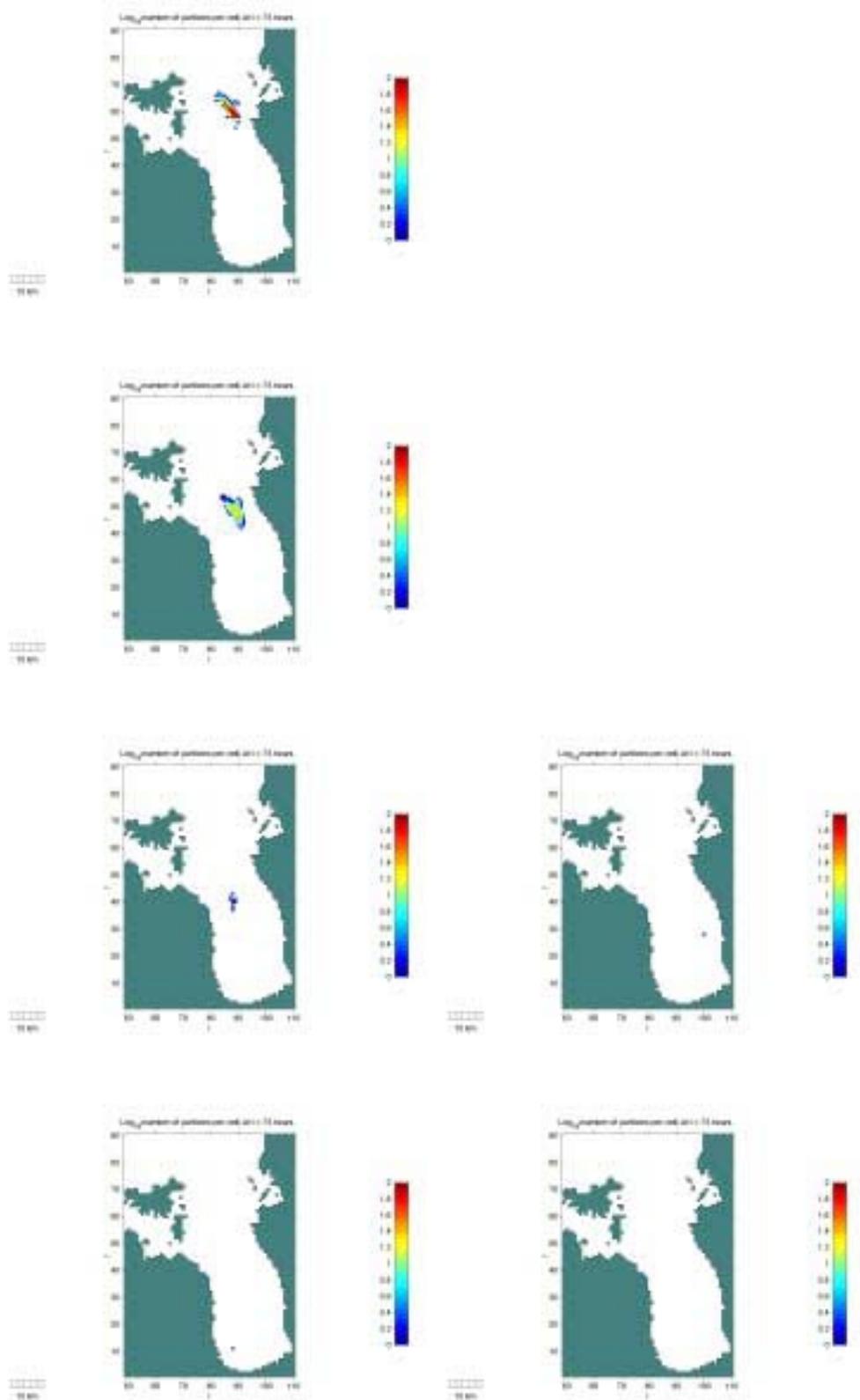
Figure 26.

Depth-averaged water-column concentration (plotted as Log_{10}) of a 0.02 m s^{-1} settling-tracer five tidal cycles after release at sites 1 to 6 under calm conditions into a stratified water column. Plot layout follows release site location (Figure 1). Tracer that has settled onto the seabed is not shown – hence the absence of any particles in the bottom two illustrations. If all the tracer were contained in one cell, it would have a Log_{10} concentration of ~3.6.
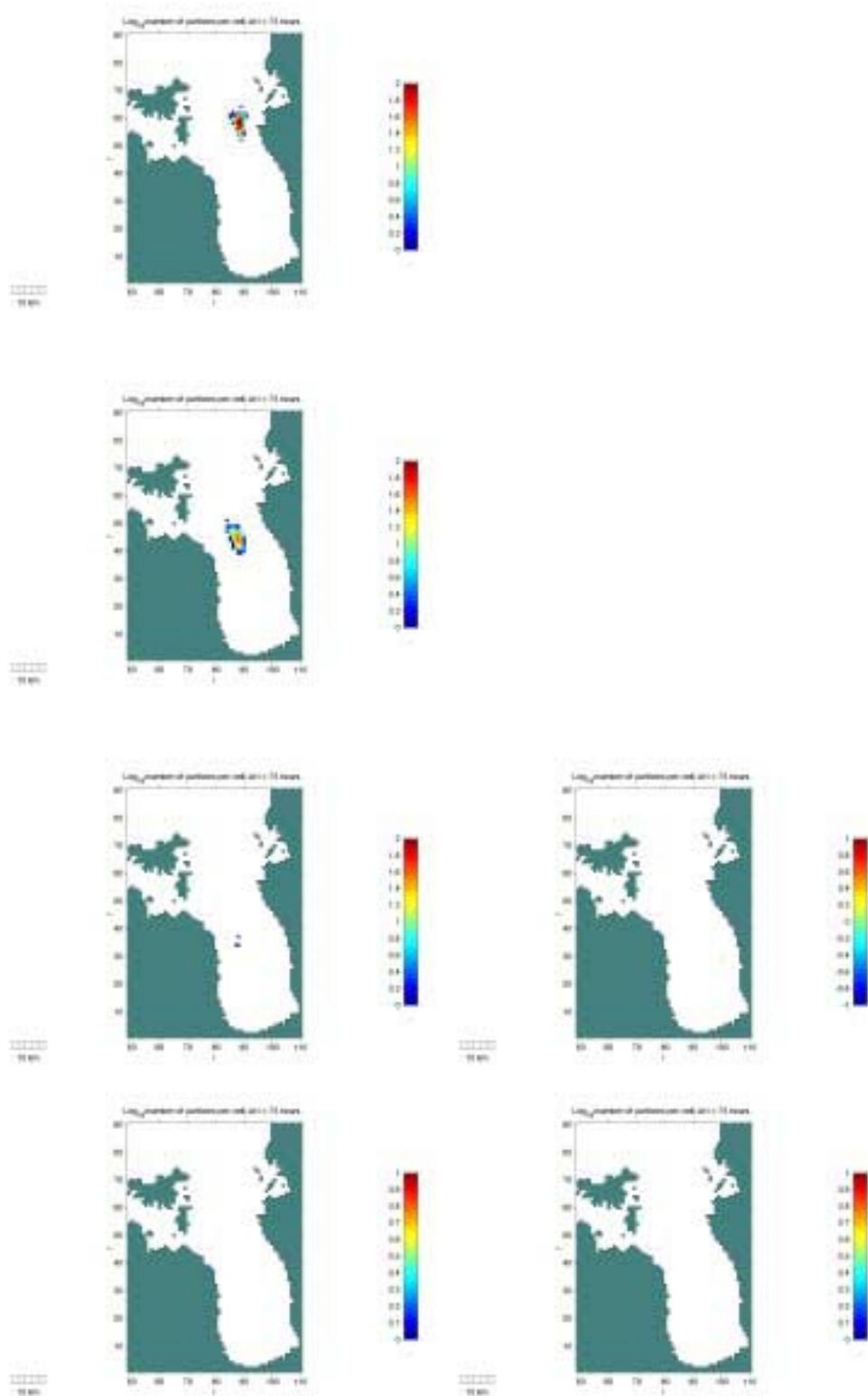
Figure 27.

Depth-averaged water-column concentration (plotted as Log_{10}) of a 0.02 m s^{-1} settling-tracer five tidal cycles after release at sites 1 to 6 under a 15 m s^{-1} NE wind into a stratified water column. Plot layout follows release site location (Figure 1). Tracer that has settled onto the seabed is not shown. If all of the tracer were contained in one cell, it would have a Log_{10} concentration of ~3.6
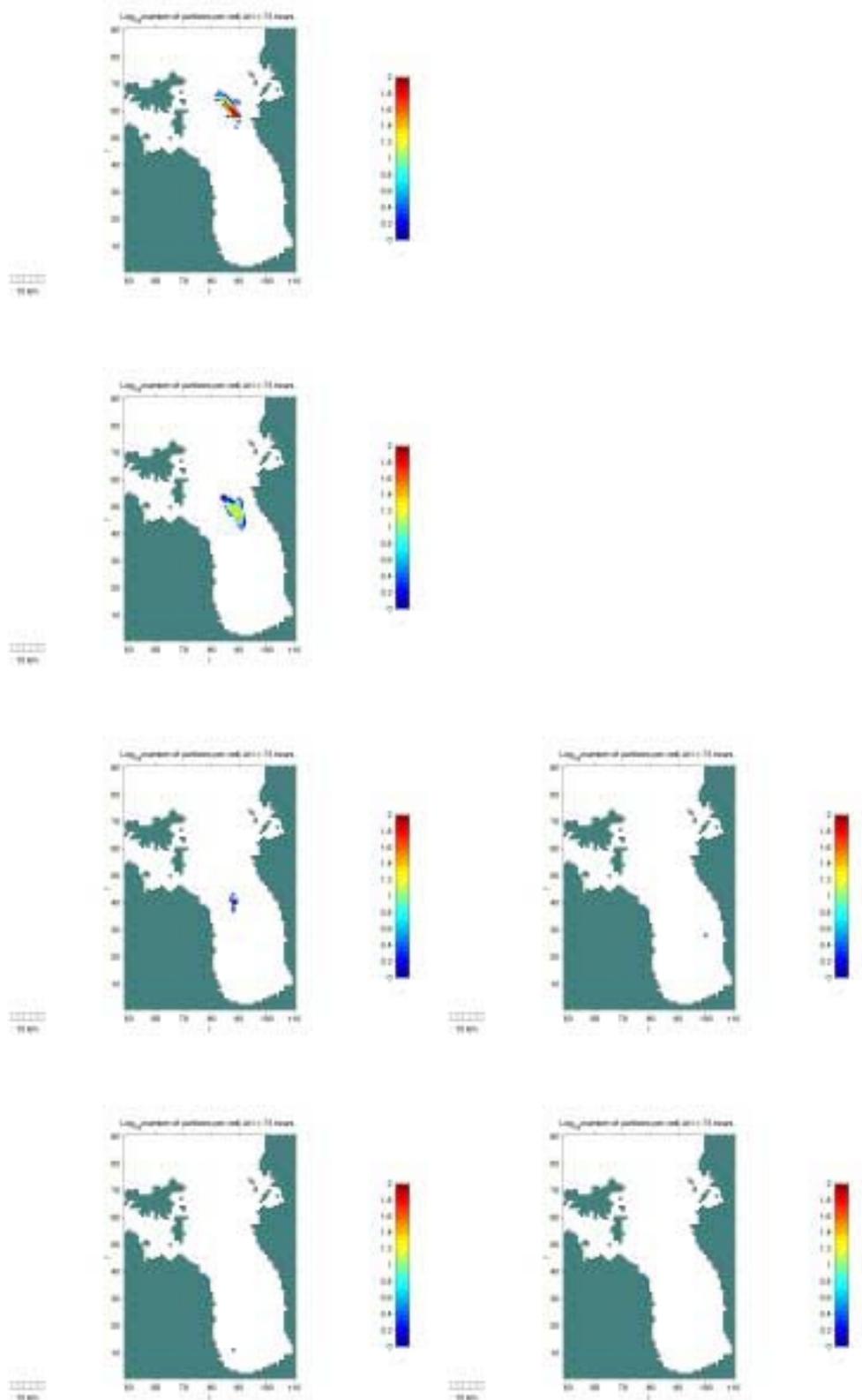
Figure 28.

Depth-averaged water-column concentration (plotted as Log_{10}) of a 0.02 m s^{-1} settling-tracer five tidal cycles after release at sites 1 to 6 under a 15 m s^{-1} SE wind into a stratified water column. Plot layout follows release site location (Figure 1). Tracer that has settled onto the seabed is not shown. If all the tracer were contained in one cell, it would have a Log_{10} concentration of ~3.6.
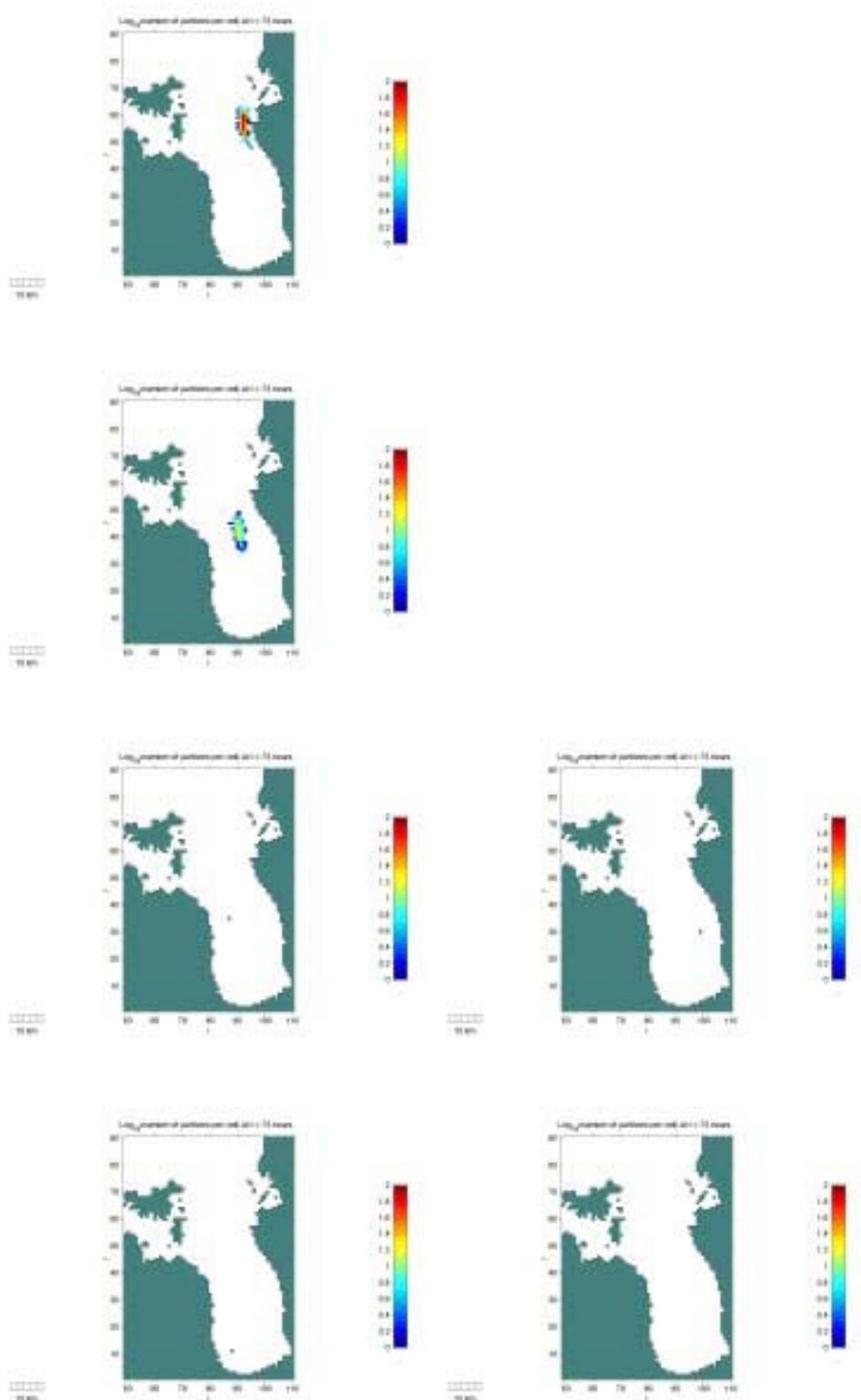
Figure 29.

Depth-averaged water-column concentration (plotted as Log_{10}) of a 0.02 m s^{-1} settling-tracer five tidal cycles after release at sites 1 to 6 under a 15 m s^{-1} SW wind into a stratified water column. Plot layout follows release site location (Figure 1). Tracer that has settled onto the seabed is not shown. If all the tracer were contained in one cell, it would have a Log_{10} concentration of ~3.6.
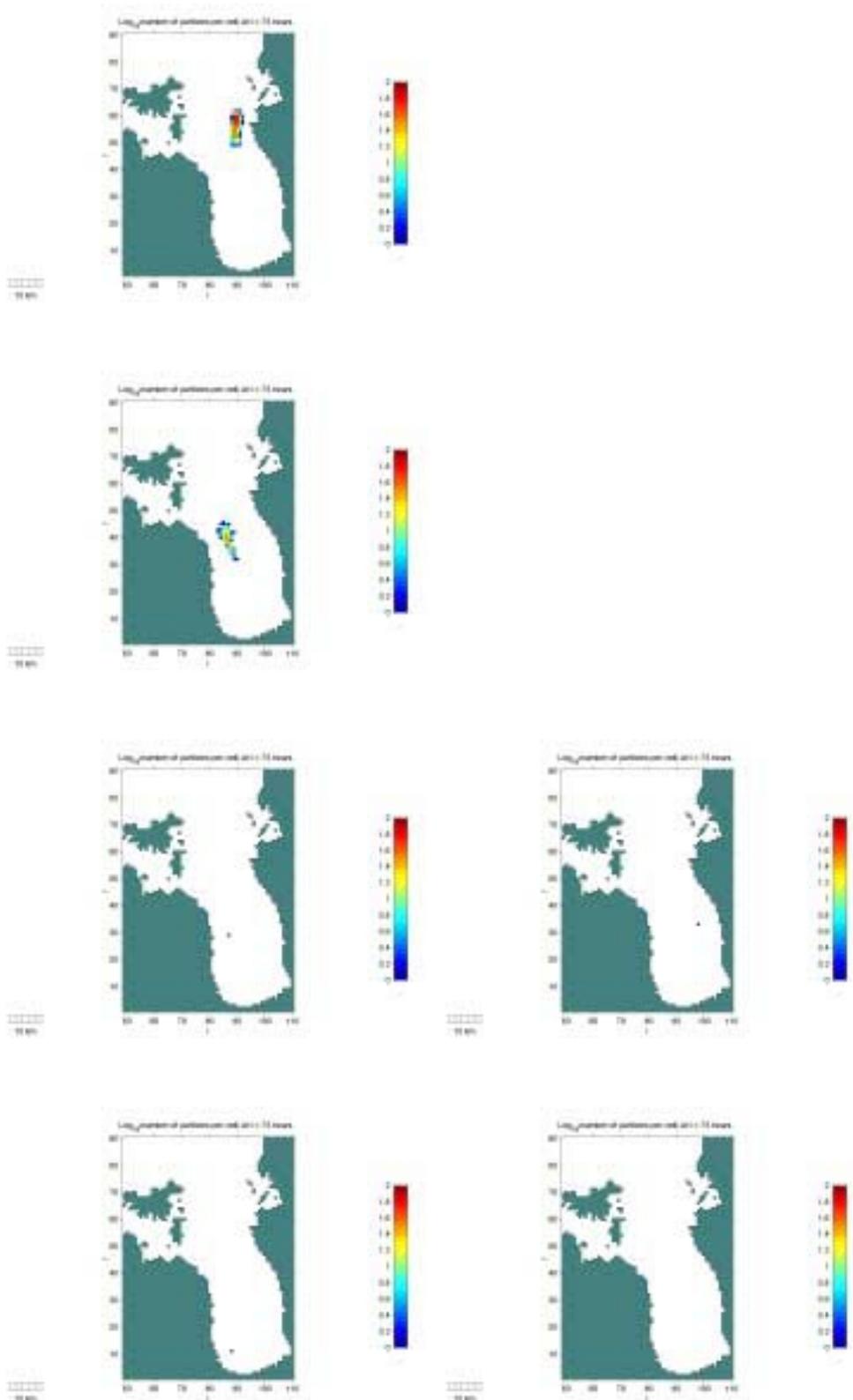
Figure 30.

Depth-averaged water-column concentration ($\log_{10}$ scale) of a 0.02 m s^{-1} settling-tracer five tidal cycles after release at sites 1 to 6 under a 15 m s^{-1} NW wind into a stratified water column. Plot layout follows release site location (Figure 1). Tracer that has settled onto the seabed is not shown. If all the tracer were contained in one cell, it would have a $\log_{10}$ concentration of ~3.6.
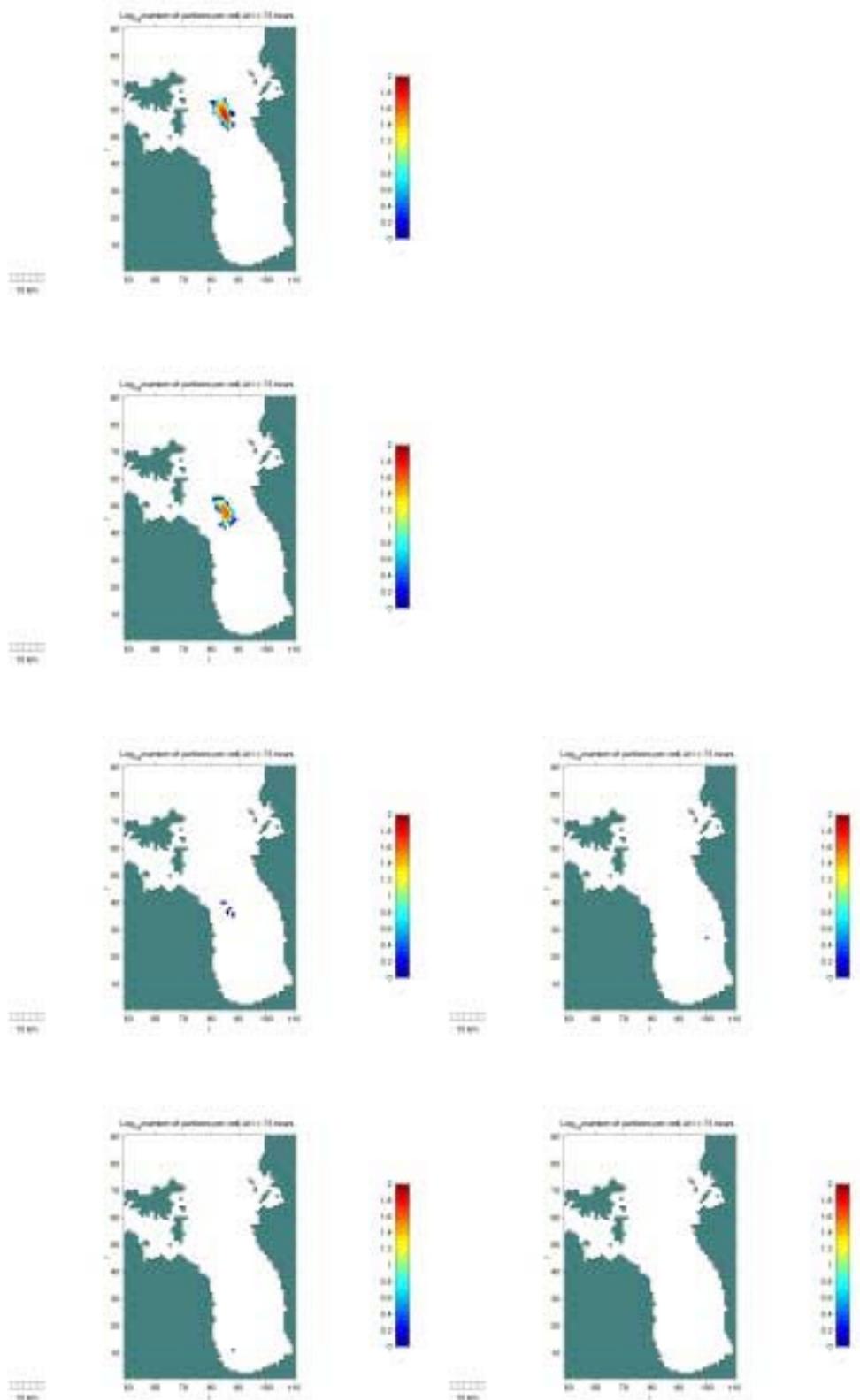
5.3.3.2 Inferences for biologically active materials

The obvious conclusion from the particle tracing work is that material disperses more rapidly from northerly release locations than it does from southerly ones; however during periods when wind-speeds are low, dispersal rates will be very low even in the northern reaches of the Firth. The corollary to this latitudinal dependence is that, for a given total area of farm within the Firth, any effects are likely to be locally less severe (but, spatially more extensive) at more northerly sites (because the water which passes through the farms is quickly mixed with pristine water from elsewhere). It would, however be incorrect to conclude that the far-field effects from a farm will extend no further than (or even, as far as) indicated by these figures.   On the one hand, materials will continue to disperse beyond our simulated five-tidal cycles (taking them further afield). On the other hand, none of the materials likely to be consumed / released by a farm are entirely biologically inert, thus regenerative processes may be important (these are absent in this dispersal model) . Thus, the farm footprint for a particular material will depend not only upon the rate at which it is diluted and transported away from the farm (indicated by our tracer studies), but also upon the intrinsic rate at which this material is regenerated through biological processes (a material-specific characteristic). Nutrients, phytoplankton and protozoa are highly reactive (high rates of nutrient uptake/excretion, high growth rates). Thus, in contrast to the situation with our inert tracer, recovery of these materials is likely to be dominated by growth processes rather than recolonization/dilution processes. To make quantitative predictions of the size of a farm's footprint for these reactive materials, we need a more sophisticated model which takes explicit account of nutrient cycling processes and growth by phytoplankton/microzooplankton (see section 7.2 for a biophysical model which takes explict account of nutrient-phytoplankton interactions). Nonetheless, by comparison of the simulation-derived tracer transport speeds with documented growth rates for reactive species, we can make some semi-quantitative predictions.

Maximal weight-specific growth rates (r) for phytoplankton and protozoa (one component of the microzooplankton) are usually in the range 0.5 to 2 d^{-1} (ie an individual takes between 0.5 and 2 days to double its weight). Most of these organisms reproduce by binary fission (i.e. one large cell splits into two, smaller 'daughter' cells). Thus, generation times are also in the range 0.5 to 2 days. A crude estimate of a farm's footprint for these highly reactive species can be gained by first determining the time (T, days) which would be required for the residual population of size "% depletion" (relative to its abundance prior to passing into the farm) to regenerate to the size at which it entered the farm:

$$T \approx \frac{1}{r} \log_e \left(\frac{100}{\% \text{ depletion}} \right)$$

and then determining how far beyond the farm the water would have travelled in this time. For example, a population which suffers 50% depletion through a farm, and subsequently grows at a rate of 1 d^{-1}, will take approximately 0.3 d to recover to its pre-impact level (neglecting immigrants from pristine water). If the current velocity is 5 cm s^{-1}, it will travel approximately 1.3 km in this time.

Conditions which slow individual, or population growth rate (for phytoplankton: low light, low nutrient concentrations, numerous grazers; for microzooplankton: little food, numerous predators) will operate to increase the spatial extent of a farm's footprint. Mussels consume phytoplankton, but produce inorganic nutrients. Thus, immediately downstream of a farm conditions may be more favourable to the growth of phytoplankton (which require the nutrients) than they are for zooplankton (which require the phytoplankton). This implies that, even for fast-growing zooplankton, a farm's zooplankton footprint may prove to be larger than its phytoplankton one. High current velocities also have the potential to increase the spatial extent of any footprint, but they will also reduce the initial degree of depletion.

Larger zooplankton (which have generation times of a week or more) can be considered to be almost inert over the time-scales required to disperse throughout the Firth of Thames. Furthermore, the eggs/larvae of benthic organisms and fish are almost perfectly inert over these time-scales (because the generation times of these organisms are months to years in duration). Thus, we argue that our tracer analysis provides strong evidence that, for slow growing organisms, wind driven circulation patterns will extend the far-field influence of a farm over the entire Firth of Thames within a matter of weeks. Whether or not the far-field influence can be detected will depend upon how much dilution has taken place during this period. This will depend upon the quantity of pristine water in the Firth, which for relatively inert materials such as larger zooplankton is likely to decline in proportion to the total quantity of mussel aquaculture within the Firth and Hauraki Gulf.

As noted previously, recovery may also be driven by dilution of impacted waters with pristine waters. We have made a preliminary estimate of the importance of transport/mixing in promoting recovery around farms (Table 6); however we emphasize that these values are valid only when the inflowing water is pristine. In situations where the inflowing water is impacted (because it contains residual contamination from this, or another farm), mixing will be much less effective in promoting recovery (because the 'dilution' effect is less marked). Theoretically, our particle-tracking model could be used to make quantitative predictions of the footprints of individual farms and of the cumulative footprints of wide-spread aquaculture development under circumstances where the inflowing water is not pristine, but this would require that the model be run for a much longer period of time, and that particles be released throughout the simulation period. It was not possible to undertake an exercise of this magnitude within the present project. Furthermore, we believe that the results will be highly dependent

upon: (a) the size of individual farms, (b) the absolute distribution of farms, and (c) the relative distribution of farms since these factors will determine the relative abundance of pristine and impacted water, and the extent to which they mix with one another. Thus, in addition to substantial computing resources, such an analysis requires a tightly specified suite of scenarios.

One interesting implication of the "recovery-by-mixing" process is that farm design might have a substantial role to play in determining the spatial extent of a farm's footprint. For example, the surface area for mixing exchange between impacted and pristine waters could be increased by sub-dividing a farm into numerous sub-farms, separated by **sufficiently large** areas of open water (at the expense of increasing the total area of sea-bed enclosed within the bounds of the whole farm). Intuitively, it is reasonable to expect that this arrangement would be most effective in promoting rapid recovery when the lines of clear water run parallel with the prevailing residual current, and when the long axis of a farm runs perpendicular to this axis. At present, it is not possible to progress from these intuitive statements toward more quantitative ones.

5.4 Flushing Times within the Firth of Thames

NIWA has developed a simple model to predict the relative degree of phytoplankton depletion downstream of a mussel farm as a function of several variates – including the flushing time of the farm. Within this sub-section, we aim to derive preliminary estimates of the flushing time at each of the six locations used for the particle-tracking simulations. In section 6 (which follows immediately afterwards), we apply these within the simple model of depletion in order to derive preliminary estimates of the extents to which phytoplankton may become depleted within different parts of the Firth of Thames.

The flushing time is generally defined as the ratio of the volume of a particular body of water to the net rate at which water passes through this volume. Perhaps the simplest interpretation of the flushing time is to consider it as the 'time which would be required to refill the volume in question – were it to become emptied'. Thus, if a region has a short flushing time, the water within it is rapidly replaced with water from elsewhere. Conversely, if it has a large flushing time, the water is replaced only slowly. Thus, the flushing time is one characteristic of the local mixing rate. If the influent water is pristine, then the flushing time (or rather, its inverse) provides an indication of the rate at which any effects associated with local aquaculture will be 'diluted' through transport processes. In an area with a short flushing time, the effects will tend to be **locally less intense**, but **spatially more extensive**.

Since flushing time is one measure of the local mixing rate, the ratio of an area's flushing rate (inverse of flushing time) to in-situ rates of biological processing provides a means of determining the relative importance of: (a) transport processes (dilution with imported

'pristine' water and export of 'contaminated' water, and (b) in-situ 'biological recovery' processes as mediators of the ecological health of the local system. Aquaculture-induced deviations from the locality's production and ecological carrying capacities are much more likely in regions where cleansing are dominated by in-situ processes.

Despite the intuitive appeal of the flushing-time characteristic, it is important to understand that it is spatial-scale dependent. The flushing time depends upon the speed at which small parcels of water move through the area. It is obvious that, for a given travel-speed, the flushing time will increase as the size of the area in question increases. This raises the question: at what scale should we attempt to define a flushing time? There are two obvious candidate scales: the farm-scale and the 'administrative-area', or 'zone-scale' (ie the scale at which Regional Councils might choose to regulate aquaculture operations). For the purposes of this report, we have chosen to define the flushing time as the 'time-taken-to-traverse-750 m' (the horizontal resolution of the hydrodynamic model of the Firth of Thames). This scale is, perhaps a little larger than most extant farms (though smaller than many proposed ones). It is also consistent with regulation at a 'bay-scale', but much finer than would be consistent with regulation on a coastal-zone scale.

In the Firth of Thames, and any other region where current velocities (ie speed and direction) are variable, there is an added complication: the flushing time will depend upon the prevailing currents during the period over which the flushing time is calculated. For example, consider a small area through which tidal movements alternately push and pull the water. At a sub-tidal time-scale, water may pass through the area in a matter of tens of minutes, but at a super-tidal time-scale, individual parcels of water may show little net displacement. In this situation, whilst a farm's impacts would be distributed over a volume larger than the farm itself, they would nonetheless remain within a (nearly closed) area. Given the continuous nature of aquaculture effects, we argue that the flushing time should be calculated on the basis of a super-tidal time-scale. Consequently, we use the residual current speed (ie net travel speed over one tidal cycle) when calculating the flushing time.

Even after defining the flushing time on the basis of tidal residuals, an area's flushing time remains time-scale dependent – and hence dependent upon prevailing wind conditions. This weather-dependence is further exemplified in Table 7, which lists residual near-surface velocities for the 6 areas used in the preceding particle-tracking analysis. The average velocity from all sites during calm conditions was 0.015 m s^{-1}, while the average from all sites and the four 15 m s^{-1} winds was 0.047 m s^{-1}. This re-emphasises the importance of wind in driving circulation patterns within the Firth of Thames. Estimates of the flushing time for one model grid cell (the estimated mean time taken for a tracer particle to traverse one 750 m cell) are also calculated in Table 7. The shallow (1-3 m) sites in the southern Firth of Thames have slow residual currents and correspondingly long flushing times. Residual current speeds increase to the north

and flushing times are correspondingly reduced. The overall mean and vector-averaged-mean cell flushing times from all simulations were 3.6 and 31.6 hours respectively (the former is based upon the mean water-**speed** over a tidal cycle, whilst the latter is based upon the mean **velocity** over a tidal cycle; from the point of view of this analysis, the latter is a superior measure because it takes better account of the fact that water may slosh back and forth during the cycle such that though the instantaneous current speeds are high, the longer-term speed is low).

Table 7:

Residual near-surface (0-2 m) velocities (m s^{-1}) extracted from hydrodynamic simulations by (Oldman & Senior 2000) at sites 1 to 6. Mean flushing times for one model grid cell are also calculated, i.e. the estimated mean time taken for a tracer particle to traverse one 750 m cell.   Note, the final column is a simple average that takes no account of the relative frequencies of the five different wind directions.

Site	Calm	NE	SE	SW	NW	Site Mean	Flushing (hours)	Site Mean (vector-averaged)	Vector-averaged flushing (hours)
1	0.009	0.020	0.010	0.037	0.013	0.016	12.7	0.009	23.0
2	0.038	0.078	0.030	0.010	0.052	0.063	3.3	0.039	5.4
3	0.003	0.019	0.017	0.011	0.012	0.017	12.3	0.004	55.1
4	0.005	0.032	0.029	0.027	0.033	0.028	7.4	0.007	30.1
5	0.025	0.081	0.092	0.083	0.061	0.081	2.6	0.018	11.5
6	0.011	0.093	0.102	0.099	0.086	0.088	2.4	0.012	17.6

The flushing times that we have calculated are minima. This is because they are calculated using horizontal current speeds from the hydrodynamic model's surface-most layer (0 – 2 m) – where the (wind-driven) residual currents are largest. Furthermore, in deriving them, we used a wind-speed of 15 m s^{-1}, which is above average for the Firth of Thames (Figure 5). At the other extreme, Zeldis & Smith (1999) calculated a flushing time of approximately 56 days for the Hauraki Gulf as a whole (incl. the Firth of Thames).

The flushing times that we have derived are similar to those inferred at the **farm-scale** (ie, somewhat smaller spatial scale) within the Marlborough Sounds (A.H. Ross, *pers. comm*).   Correcting for the observation that most farms in the Marlborough sounds are <750 m x 750 m in horizontal extent, this implies that, in the northern parts of the Firth of Thames, transport-cleansing is likely to be relatively more important than it is within the Marlborough Sounds. Note however, that this statement assumes: (a) farms of

similar size, and (b) that the replacement water is, indeed, pristine. Given the size of some of the proposed farms, we know that condition (a) will not be satisfied in all cases. At present, we cannot determine at what density of farms condition (b) would be violated.

Inspection of Figure 19 and Figure 20 suggests that wind-driven residual currents in the surface waters of the northern Firth of Thames will often be in the range 5 – 15 cm s^{-1} (for a relatively high wind-speed of 15 m s^{-1}). A combination of observational work within- and simulation studies of Beatrix Bay led Inglis *et al.* (2000) to define five velocity regimes with respect to mussel aquaculture,: <5 cm s^{-1}, 5-10 cm s^{-1}, 10-20 cm s^{-1}, >20 cm s^{-1}. The authors argue that, given the farm-sizes typical of the Marlborough Sounds, areas with current speeds of <10 cm s^{-1} are likely to suffer phytoplankton depletion both within- and immediately downstream of the farm. Areas with current speeds of 10-20 cm s^{-1} are likely to exhibit appreciable depletion only downstream of the farm, and depletion may not be observed at all in areas where current speeds exceed 20 cm s^{-1}. Comparison of these five ranges with simulated residual currents within the Firth of Thames suggests that it is probably that both within- and downstream- phytoplankton depletion will be observed if the Firth farms have similar characteritistics (size, line density etc.) to those of the Marlborough Sounds. Furthermore, unless line densities, or mussel densities per line are correspondingly reduced within the very large farms proposed for the Firth of Thames, it is likely that, for any given velocity, depletion will be more extreme downstream of the very large farms (in comparison with those within the Marlborough Sounds).

₆ Depletion Estimates

6.1 Introduction

Shellfish filter large quantities of suspended organic and inorganic material from the water column. They subsequently ingest a portion of the organic material, but some (perhaps most) is ejected (along with the inorganic material) as pseudofaeces.   The pseudofaeces is enclosed within a mucal coating, and sinks at speeds of circa 2 cm s⁻¹ (N. Hartstein *pers. comm.*). Thus, any organism which is entrapped during the initial filtration is removed from the water-column- regardless of whether or not it is ingested by the organism.

It is known that *Perna canaliculus* can efficiently entrap organisms as small as 5 μm, and as large as 100 μm (James et al. 2001). Less formal observations also suggest that they are able to consume particles upto a size of 200 μm (M.R. James, pers. obs.) but we do not know whether they are also able to entrap organisms larger than 200 μm or smaller than 5 μm (studies on the Blue mussel *Mytilus edulis* have shown that it is able to filter organisms as small as 2 μm, and perhaps even smaller (Bayne et al. 1976)). It is conceivable that *P. canaliculus* may be able to remove any planktonic organism which is larger than 5 μm and not sufficiently mobile to avoid capture. If this is the case, they will remove not only the majority of phytoplankton and protozoa (which fall in the 5 – 100 μm size range), but also many mesozooplanktonic organisms, and perhaps even the eggs and early larval stages of fish and macro-benthic organisms – as the blue mussel, *Mytilus edulis*, has been shown to do (Davenport et al. 2000).

It is not merely a conceptual possibility that mussel farms may cause substantial local depletion of plankton. It has been possible to detect depletion within operating farms (Grange & Cole 1997; Ogilvie et al. 2000).

Within this section of the report, we use a simple model which incorporates experimental measurements of (mussel) size-dependent filtering rates (e.g. James *et al.* (2001)), population size structure of mussels within a farm, farm size (total number of long-lines) and flushing time to derive estimates of the degree of depletion immediately downstream of a farm (i.e. at the downstream boundary of the farm). We make these calculations for farms of varying intensity (long-lines ha⁻¹) at six nominal sites within the Firth of Thames. These correspond to the size sites used in the preceding section. For the purposes of our calculations we make the following assumptions:

1. the mussel population is divided into 4 size fractions (20-35 mm, 35-60 mm, 60-85 mm, 85-110 mm). The smaller size-class will be referred to as 'spat', the latter three size classes will be referred to as 'crop'.

2. The densities (mussels m^{-1} of dropper lines) of these four size fractions are assumed to be 1000, 180, 150 and 120 respectively.

3. 20% of the total long-line length is assumed to be devoted to spat production. The remaining 80% is assumed to be devoted to crop (with equal proportions of this 80% being devoted to the three size fractions of crop mussel).

4. The ratios of dropper line-length to long-line length is assumed to be 60 for spat-lines and 24 for crop lines.

5. Based upon values typical of the Marlborough Sounds, we assume a long-line density of 1.5 lines ha^{-1}, each of 110 m length.

6. We vary the farming intensity within our notional 56.25 ha farm by increasing or decreasing the number of long-lines within the 56.25 ha. Thus, at an intensity of 100%, the number of long-lines is 1.5x56.25 ($\cong$ 84) lines per 56.25 ha, and at an intensity of 10% this falls to $\cong$ 8.4 lines in the 56.25 ha.

7. Location-specific flushing times are derived from the residual velocities predicted by the hydrodynamic model of the preceding section.

8. Mussel size-specific volumetric filtration rates (L mussel^{-1} d^{-1}) are derived from literature values (e.g. Hawkins et al. 1999).

The model is described in more detail within Appendix 3. We emphasize that the model makes numerous simplifying assumptions. Thus, though it purports to predict the extent (%) to which a population of phytoplankton will become depleted as it passes through a farm, these percentages should be not be interpreted too rigidly. Indeed, rather than being predictions of the absolute degree of depletion, the percentages may be better viewed as providing an indication of the probability with which some (detectable) depletion may occur.

6.2   Results

Table 8 lists the predicted extents to which the phytoplankton population passing through the farms would become depleted for a range of farm intensities at our six nominal locations. Not surprisingly the magnitude of depletion is inversely related to the flushing time (which depends upon average water speeds (Figures 19, 20) and local water depth (Figure 1)) and predicted to become more severe as the farm intensity increases. Similarly, depletion is predicted to be more severe in the south of the Firth – where current velocities are lower. At the maximum farming intensity considered, it is predicted that the mussels would consume between 10% (site 2) and 120% (site 3) of the phytoplankton which pass through the farm (see sub-section 6.3 for an explanation of how depletion can exceed 100%). Farms operating at 10% of this maximum are

predicted to consume between 1% (site 2) and 12% (site 3) of the phytoplankton which pass through the farm.

Table 9 is based upon the same calculations, but in this case we express the farms' phytoplankton demand as a fraction of the total phytoplankton population resident within the 56.25 ha grid-cell in which each farm is located. This is in contrast to Table 8, which records the extent to which the water immediately downstream of the farm is depleted. In effect, Table 9 provides a simple means of addressing the question: "…rather than considering the maximum depletion which will occur (ie that which we have predicted at the downstream end of the farm), what is the average depletion across the entire 56.25 ha of the farm?". Specifically, we are asking: at a scale of 56.25 ha, how severe would a mussel farm's impact be if its demand were mixed through the entire 56.25 ha in one day? Given the flushing times of our six chosen areas (Table 7), we know that, in many cases, the farm's load will be transmitted into a volume less than this over the course of a day – that is, though the farm is spread over 56.25 ha, depletion is likely to be more tightly restricted around individual long-lines. This is particularly true when the line density is low. Nonetheless, we argue that the figures in Table 9 provide **crude** indications of the larger-spatial scale significance of any depletion.

Table 8.

Predicted degrees of phytoplankton depletion immediately downstream of a farm sited at locations 1-6 (Figure 1). For each site, several depletion figures are quoted. These correspond to farms of differing intensities (as a percentage of the default 1.5 lines ha^{-1}) within the grid cell containing the nominal site. See text for explanation of how the predicted depletion can exceed 100%.

Site	1	2	3	4	5	6
Flushing time for the grid-cell (h)	23	5.4	55.1	30.1	11.5	17.6
Farm intensity	% Depletion	% Depletion	% Depletion	% Depletion	% Depletion	% Depletion
100%	50.30%	11.81%	120.50%	65.83%	25.15%	38.49%
80%	40.24%	9.45%	96.40%	52.66%	20.12%	30.79%
60%	30.18%	7.09%	72.30%	39.50%	15.09%	23.09%
40%	20.12%	4.72%	48.20%	26.33%	10.06%	15.40%
20%	10.06%	2.36%	24.10%	13.17%	5.03%	7.70%
10%	5.03%	1.18%	12.05%	6.58%	2.52%	3.85%
5%	2.52%	0.59%	6.03%	3.29%	1.26%	1.92%
2%	1.01%	0.24%	2.41%	1.32%	0.50%	0.77%
1%	0.50%	0.12%	1.21%	0.66%	0.25%	0.38%
0.1%	0.05%	0.01%	0.12%	0.07%	0.03%	0.04%

Table 9.
Predicted degrees of phytoplankton depletion within the 56.25 ha grid cell containing the farms listed in Table 8. In this case, we calculate the depletion as a fraction of the total phytoplankton population within the 56.25 ha of the grid-cell, rather than as a fraction of the population of phytoplankton which passes through the farm.

Site	1	2	3	4	5	6
Flushing intensity (h)	23	5.4	55.1	30.1	11.5	17.6
Farm area	Depletion	Depletion	Depletion	Depletion	Depletion	Depletion
100%	2.10%	2.10%	5.00%	2.74%	1.05%	1.60%
80%	1.68%	1.68%	4.00%	2.19%	0.84%	1.28%
60%	1.26%	1.26%	3.00%	1.65%	0.63%	0.96%
40%	0.84%	0.84%	2.00%	1.10%	0.42%	0.64%
20%	0.42%	0.42%	1.00%	0.55%	0.21%	0.32%
10%	0.21%	0.21%	0.50%	0.27%	0.10%	0.16%
5%	0.10%	0.11%	0.25%	0.14%	0.05%	0.08%
2%	0.04%	0.04%	0.10%	0.05%	0.02%	0.03%
1%	0.02%	0.02%	0.05%	0.03%	0.01%	0.02%
0.1%	0.00%	0.00%	0.01%	0.00%	0.00%	0.00%

6.3 Discussion

For reasons which will be made more clear in the following paragraph, we argue that the estimates of depletion (Tables 8 & 9) are, at best semi-quantitative. We suggest that they are better understood to be indicative of the risk that depletion may occur rather than being accurate predictions of the likely degree of depletion that will occur. In particular, they are useful in determining whether a farm of a given size is more likely to cause depletion in one site than it is at another site.

The reader may wonder how realistic the predictions of Table 8 and Table 9 are – particularly in the light of one prediction that depletion will exceed 100% (this is physically impossible because it implies that the shellfish remove **more than** all of the phytoplankton which pass through the farm!). This arises as a consequence of the simple nature of the model. The most dubious simplifying assumption made in deriving the depletion model is that there is no overlap of the volumes of water that each mussel filters. In reality, most mussels probably re-filter at least some of the water which has already been filtered by their near-neighbours. Indeed, it is also likely that re-filtration takes place at greater distances – with 'downstream' mussels reworking the water which was previously filtered by mussels far upstream. Experiments are presently underway to estimate the true extent of re-filtration, but it is this absence of re-filtration effects which allows our simple model to predict depletion in excess of 100%. In this sense, the model's predictions must be regarded as 'worst-case scenarios'. Re-filtration effects undoubtedly operate to reduce the extent of depletion (because the downstream shellfish cannot remove phytoplankton which have already been removed by its upstream neighbours). On the other hand, we have based our depletion calculations upon flushing times calculated from an average of five wind scenarios (calm and 15 m s^{-1}) from 4 directions. As noted previously, 15 m s^{-1} is an above average wind-speed, and our estimates of the flushing times at the six chosen sites will be shorter than the true, long term averages. In turn, this will tend to compensate (to an unknown degree) for the absence of re-filtration effects.

The question: "what degree of phytoplankton depletion is ecologically acceptable?" is difficult to answer. We suggest that it is best addressed not by consideration of percentage depletion *per se*, but rather by reference to the resultant phytoplankton concentrations. After making this transformation, it becomes possible to compare predicted concentrations with data such as those in Figure 15. For example, one might argue that phytoplankton concentrations immediately downstream of a farm should not fall below the minimum in time-series such as those of Figure 15 and determine the allowable percentage depletion accordingly.

Alternatively, the minimum allowable phytoplankton concentration might be set by reference to data concerning the algal densities that various other taxa require in order to maintain their growth or feeding rates above some specified fraction of their respective maxima. For marine copepods and protozoa, there is an extensive literature concerning these relationships, however there are fewer data for other important planktonic groups – including planktonic larval stages of benthic organisms and fish. A complete review of the literature is beyond the scope of this report, but we will offer several relevant examples.

Peters & Downing (1984) suggest that, for cladoceran zooplankton, growth will become negative (ie individuals will lose weight) when phytoplankton concentrations fall below circa 50-80 μg C L^{-1} (~ 1 μg Chl L^{-1}: assuming a C:chlorophyll ratio of ~ 50:1 (EPA 1985)). If so, one would expect to observe substantial changes in the cladoceran community if mussel farms reduced phytoplankton concentrations to below this level over substantial areas. Indeed, substantial changes may occur even at concentrations greater than this because time-to-adulthood (and, by implication, fraction of eggs reaching adulthood) and adult reproductive rate are both influenced by food density. Such changes need not be restricted to the cladocera: for example Frost (1975) reported that the calanoid copepod *Calanus pacificus* rapidly reduces the rate at which it filters water as food concentrations fall below ~ 10 μg C L^{-1} (~ 0.2 μg Chl L^{-1}). Reeve & Walter (1977) report that the small neritic copepod *Acartia tonsa* requires still higher food densities (~ 1.0 μg Chl L^{-1}) in order to maintain filtration activity. Neither of these two species is abundant within the Firth of Thames/Hauraki Gulf region, but Paffenhöfer & Orcutt (1986) present data for the cladoceran copeod *Penilia avirostris* – which is a dominant member of the zooplankton within the Firth of Thames during summer. These authors found that survival was optimal at food concentrations of circa 0.7 μg Chl L^{-1}. Survival was lower, and reproduction did not occur when individuals were provided with 0.07 μg Chl L^{-1}.

Hansen & Bjornsen (1997) made an extensive review of the literature concerning feeding and growth of zooplankton (cf planktonic larvae of benthic organisms and fish). They found that the half-saturation food concentration (food concentration at which an organism's ingestion rate is one half of maximal) was ~240 μg C L^{-1} (for zooplankters in the size range 2 – 2000 μm). This implies a half-saturation phytoplankton concentration of approximately 5 μg Chl L^{-1}. This is a little greater than the average for the northern Firth of Thames (which has the highest average standing stocks of phytoplankton in the greater Hauraki Gulf, Figure 13, Figure 15). When food concentrations are below the half-saturation constant for ingestion, uptake is approximately linearly dependent upon food concentration. This may imply that locally, zooplankton ingestion rates (and, by implication, growth rates) will fall linearly with the absolute extent (*cf* percentage) of phytoplankton depletion. We qualify the previous statement however with the observation that many zooplankton are omnivores (feeding upon smaller zooplankton and bacteria as well as phytoplankton). Thus the total quantity of food available to them will often exceed that indicated by chlorophyll concentrations. Nevertheless, mussels

are also omnivorous. Thus, they too will exploit many of the 'alternative' food sources possibly to the detriment of other planktivores.

Zooplankton growth and reproduction also depends upon the nature of the food (protein content etc.) as well as the total abundance of food (as measured by characteristics such as carbon, ash-free dry weight, chlorophyll, or cell numbers) (Kleppel 1993). Omnivory and biochemical complications make it difficult to make quantitative predictions of how aquaculture may modify planktonic dynamics but the data and observations do imply that, **as measured by local zooplankton growth rates**:

a. there may no lower threshold farm density below which there will be no adverse **local** impacts, and even moderate levels of depletion **may** induce local changes in the feeding, growth and reproductive rates of some of the zooplankton.

b. the spatial extent and local magnitude of such rate reductions are both likely to increase approximately linearly with the total area of farms within the Firth of Thames.

We know of no studies which have set out specifically to determine what level regional-scale, or meso-scale depletion can be sustained without adverse consequences elsewhere in the ecosystem, but Rodhouse & Roden (1987) recommended that only 50% of Killary Harbour's potential (mussel) food supply be exploited for mussel aquaculture.   Ince & Lutz (1981) adopted a similar criterion in their carrying capacity model. We do not know whether these recommendations have been followed, and if so, with what consequences. Nonetheless, given that standing stocks of phytoplankton in the Firth of Thames may not be sufficient to saturate zooplankton feeding rates, we suggest that **wide-scale** depletion of this extent would be likely to have a detectable influence upon zooplankton production within the Firth of Thames.

7 Scoping of Research Leading to an Improved Estimate of the Sustainability of Shellfish Aquaculture within the Firth of Thames

We believe that substantially improved estimates of the sustainability of shellfish aquaculture within the Firth of Thames require additional developments in several disciplines.   There are two categories of issue requiring additional information: those which are not site-specific and those which are site-specific. We will refer to the two categories as **generic** and **site-specific** respectively. Many of the former issues relate to areas of fundamental research, and we will address these after addressing the site-specific issues. Nonetheless, as will become clear, accomplishment of some of the site-specific tasks will require at least a modicum of progress in fundamental areas.

7.1   Site specific issues

We believe that the following site-specific issues need to be addressed (the order is approximately indicative of our perception of importance, and also of the ease with which the task can be achieved):

❑ Longer-term hydrodynamic simulations using observed time-series of wind to derive better estimates of the long-term tidal-residual currents throughout the Firth of Thames. These simulations are prerequisites for the three items which follow. It would be advisable to make simulations for El Niño and La Niña conditions.

❑ Long term simulations of the distribution of **continuously** released, biologically inert tracers from the locations of proposed farms to better quantify (a) to where impacts may be transported, and (b) to what extent they will be dissipated by mixing (rather than biological regeneration). This task relies upon first making the long-term hydrodynamic simulations.

❑ Similar simulations of biologically active species with-, and without mussel farms in order to quantify the extent and magnitude of individual far-field footprints and cumulative footprints. This task also relies upon first making the long-term hydrodynamic simulations.

❑ Assessment of the **Production Capacity** of the Firth of Thames under different development scenarios.

- Spatio-temporal distribution and abundance of faecal contaminants within the Firth of Thames in order to better determine whether (and if so, where) faecal contamination may preclude aquaculture activities

- Determination of the degree to which the abundance of key species of fish and benthic organisms are limited by egg/larval survival (if mussels are shown to consume either: these eggs/larvae, or the prey upon which they are dependent).

- Better quantification of the extent to which farmed mussels will increase the competitive stress suffered by other organisms which feed upon suspended, or benthic fine particulates.

- Determination of whether farms will modify flow patterns (and hence, **far-field** erosion/sedimentation patterns)

7.2 Generic issues

We suggest that the following represent important generic issues. Of these items, many are already being persued (as noted with each bullet) with FRST funding; however, additional funding may be necessary to adapt the methods/results to the Firth of Thames situation.

- Better characterization of the range organisms which *Perna canaliculus* and *Crassostrea gigas* are able to remove from the water-column. For simplicity, we will use the term 'diet' to describe this process, but the reader should remember that, because organisms which are removed from the water column may end up in pseudofaeces, we are not restricting our attention only to those organisms which are ingested. FRST-funded (CO1X0003) experiments looking at ingestion of a variety of mesozooplankton are planned for the coming year, but they are unlikely to extend as far as fish eggs and larvae or eggs and larvae of benthic organisms.

- Determination of whether any of the organisms rejected within the pseudofaeces are able to escape from the mucal coating which encases the pseudofaeces, or to survive (e.g. in an encysted form) until the pseudo-faecal particle becomes disrupted. (No funding at present)

- Incorporation of mussel farms as sinks for phytoplankton etc. and sources of nutrient / organic detritus within spatially resolved nutrient-phytoplankton models to enable us to address footprint issues for highly reactive species. The biophysical model is being developed with FRST funding (Contract CO1X0027). This model is currently at the testing/verification stage. It is hoped that this funding will also cover the costs of extending the model to represent mussel farms – though not the costs of applying it to a particular situation.

❑ Improved tools for estimating depletion within and downstream of farms. An advanced model of depletion at the rope-scale is being developed at present with FRST funding (CO1X0003). This development work is likely to require a further one- to two years.

❑ Aquaculture effects upon benthic remineralisation processes (esp. dentrification). It is hoped that a joint NIWA / University of Waikato Ph.D. student will begin to work on this issues shortly.

❑ Research into the ways in which farms modify local hydrodynamics and sediment deposition etc. Detailed measurements are being made around a relatively small farm in the Marlborough Sounds, but additional work about very large farms will also be required. This bullet is closely related to the final bullet of the preceding sub-section (Site Specific issues); however, we believe that there is a substantial amount of fundamental research to be done before it will be possible to make robust site-specific predictions.

❑ Quantifying ecologically acceptable degrees of change. No funding at present.

In the following sub-sections, we summarise the most important (generic and site-specific) tasks which would need to be undertaken within each of these disciplines.

Mussel 'Diet'

This study would require only that a number of conceptually simple experiments be conducted. In summary, mussels (and oysters) would be incubated in well-defined (species composition and size-structure) plankton cultures. Measurements of (a) the rate at which the plankton of different taxa / size are removed from the water-column (b) rate of rejection in pseudofaeces, (c) survival curves within pseudofaecal particles, would provide the requisite information.

An individual experiment would last only a few hours, but rather more time might be required to prepare, maintain and describe the requisite plankton cultures, and several (perhaps tens?) of experiments would be required to adequately characterize the diet. Care would be taken to ensure that the diet-cultures included a broad range of organisms present within the Firth of Thames: not only phytoplankton, but also microzooplankton, the eggs and nauplii of copepods, eggs and larvae of 'broadcast' (*cf* brooding) benthic organisms.

Long term simulation of inert tracers

This task would not require any new computer code developments, but would be computationally very demanding (thus, time-consuming). It would enable us to make quantitative predictions of farm footprints for relatively inert materials (eggs, and larvae etc.) would be depleted as a result of accumulated shellfish farm development. This approach would be well suited to considering the impacts of individual farms, or

determining which, of a very few prospective areas might best be zoned for marine-farming. Nonetheless, we emphasize that the computationally expensive nature of these simulations will limit the number of simulations that can be made. This implies that it is unsuitable as a tool by which to analyse numerous, alternative scenarios.

Incorporating 'static' mussel farms within a 3-d biophysical model of nutrient-phytoplankton dynamics

As has been described previously (section 5), inert tracers provide a satisfactory method by which to predict the magnitude of farm impacts upon organisms/materials which grow / recycle only slowly, however they are a very poor analogue for more rapidly growing / cycling materials. To assess the impact of farming activities upon such materials requires a model which takes explicit account of this rapid cycling. NIWA are developing such a biophysical model for the Hauraki Gulf area with FRST funding at present. The model (Broekhuizen 2000), which presently simulates the dynamics of inorganic nutrients (nitrogen, silicon), organic detritus and two groups of phytoplankton (diatoms, dinoflagellates) is a 3-dimensional extension of Broekhuizen (1999), with hydrodynamic forcing provided in the form of time-series output from an independent hydrodynamic model (Black et al. 2000). This model is being verified at present. The FRST-funded Hauraki Gulf model has a 5km resolution in the horizontal, but it is possible to run the model with 750 m horizontal resolution (though constraints of run-time and memory dictate that this may only be done for a more restricted spatial area). Figure 31 presents the results of one such simulation for the Firth of Thames.

With comparatively little effort, this biophysical model could be modified to include mussel farms as localised sources of nutrient/detritus and sinks for phytoplankton. Similarly, it could be extended to include a representation of zooplankton (though this would increase the computational burden of the model – which is already substantial, and increase the difficulties involved in calibration and verification).

In the first instance, the intent would not be to incorporate a sophisticated model of the growth of individual mussels. Rather, the model would be modified such that the user could specify the characteristics (location, size, population size structure etc.) of shellfish farms within the Firth of Thames / Hauraki Gulf. Each mussel farm would be regarded as 'static' (i.e. fixed size, stocking density, and population age structure etc.) but would withdraw phytoplankton from the local area and return nutrients and organic detritus at rates determined by the farm characteristics, local water temperatures, local phytoplankton abundance and documented behavioural physiological characteristics of mussels (e.g. James *et al.* (2001)).

This model would enable us to make an assessment of the impact of farms upon highly reactive species, but as with the particle-tracking model of inert species referred to in the preceding sub-section, it is computationally very demanding. Thus, this model

would be best suited to considering specific proposals or discriminating between a very few alternative options. It could not be used to explore numerous alternative scenarios.

Figure 31.

False colour plots of the simulated concentrations of nutrients and phytoplankton in upper 20 m of water within the Firth of Thames using the 750 m resolution variant of the Hauraki Gulf biophysical model. Top left to bottom right: dissolved inorganic nitrogen (mg N m^{-3}), dissolved reactive silicon (mg Si m^{-3}), diatoms cells (m^{-3}), and dinoflagellate cells (m^{-3}). The colour-scale corresponds to log$_{10}$ of concentration. This simulation uses simulated current fields for spring 1985 to drive transport within the biophysical model.
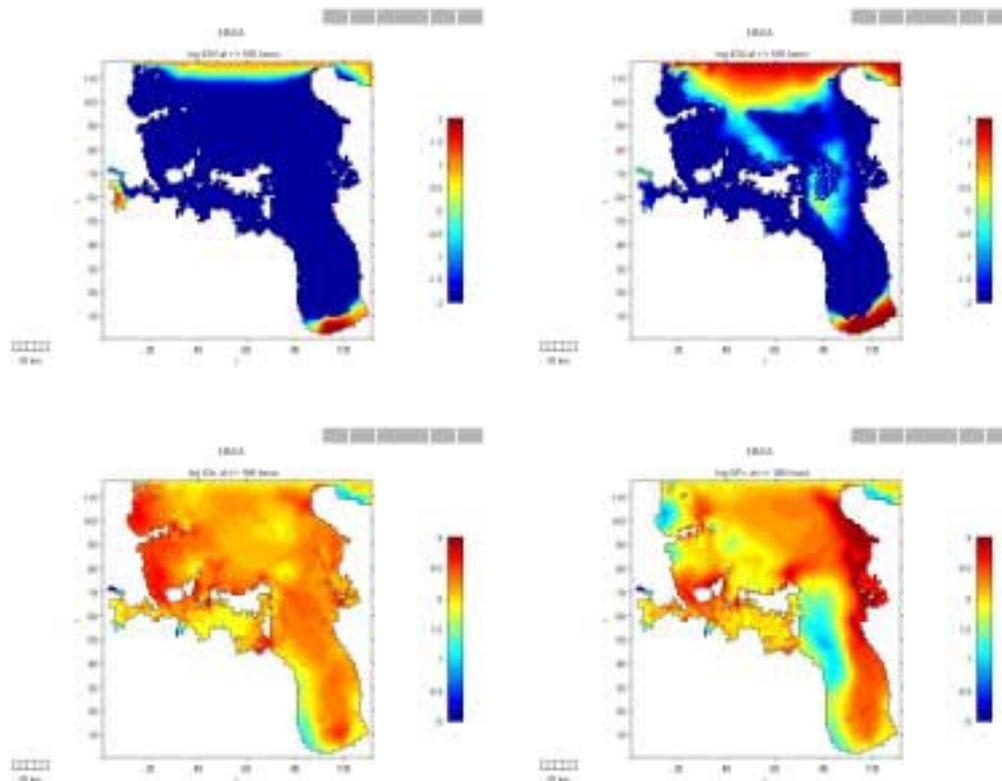

Improved estimates of local depletion

NIWA have recently begun to make detailed measurements of the manner in which mussel farms modify the local hydrodynamics (Stevens & Spigel 2001). In addition, we are exploiting new mathematical techniques (lattice gas automata) to simulate patterns of flow and nutrient / phytoplankton dynamics around individual long-lines (Figure 32). It is intended that the model will be extended to the farm-scale over the next one – two years. This will enable us to make rapid simulations of spatially detailed flow patterns, nutrient release and phytoplankton uptake around entire farms. This work is FRST funded, but would require additional funding to tailor the model to individual scenarios for the Firth of Thames.

In addition to yielding much more realistic depletion estimates than are possible at present, the highly resolved flow fields which result from this approach is also likely to provide the means by which to better predict the manners in which farms may influence wave dynamics and sediment transport/deposition.

Figure 32.

Simulations of phytoplankton abundance through cross-sections of a densely packed mussel crop line using a novel 'lattice-gas' cellular automata model. Dark green areas correspond to individual mussels, blue areas show depleted regions.
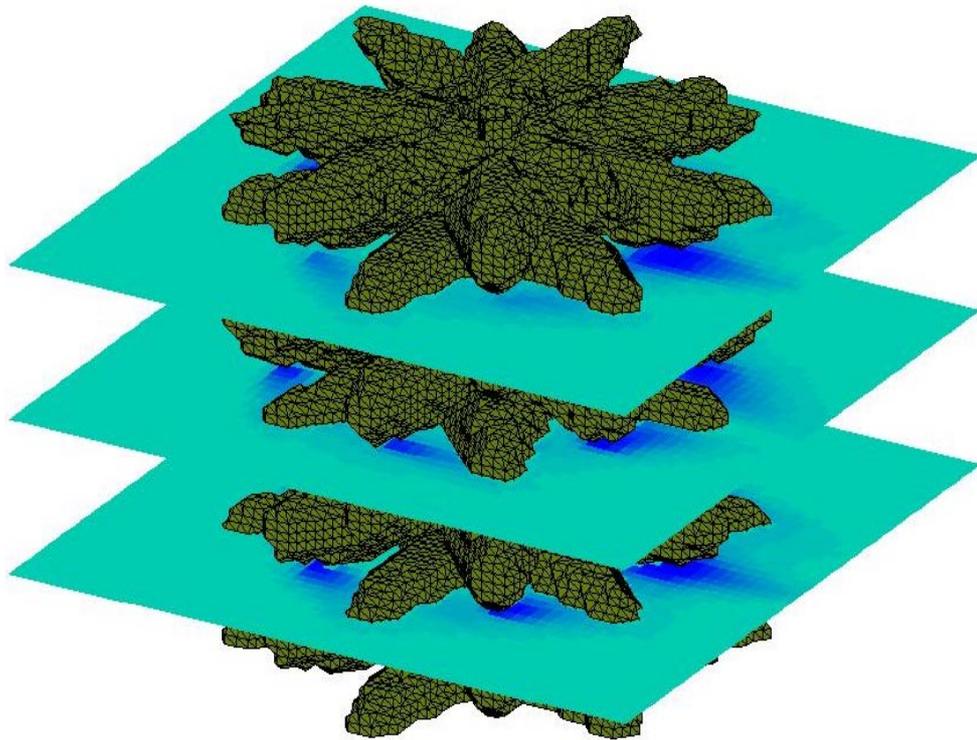
Quantifying Production Carrying Capacity within the Firth of Thames

Production Carrying Capacity is easily defined; however as noted previously, given the collective nature of the resource that is being exploited, conflict is likely to arise when individual operators are allowed to endeavour to maximise their individual yields.

With FRST funding, NIWA have developed a model which aims to address issues related to farm- and region-scale production capacity. This model couples a simplified representation of the ecosystem with a comparatively sophisticated description of mussel growth. At present, this is being applied within the Marlborough Sounds in order to determine what are the most important determinants of mussel production (Figure 33). This model could be adapted to the Firth of Thames, possibly by coupling it with the 3-dimensional biophysical model described previously. This would require substantially more development effort than would be required for the exercise of implementing

'static' mussel farms within the biophysical model which already exists for the Hauraki Gulf / Firth of Thames.

Such a model would make it possible to determine likely yields for a variety of scenarios, but given the large number of variates (farm size, farm location) it is doubtful that this approach could ever be used to identify the arrangement(s) of farms which results in a globally (*cf* locally) maximal yield.

Figure 33.

Simulations of the relationship between time-to-reach-harvestable-size and mussel stocking density for farms on the western and eastern sides of Beatrix Bay, Marlborough Sounds. The difference in the response of increased farming intensity from west to east arises because phytoplankton stocks in the western bay become increasingly depleted as the stock of western bay mussels rises. In consequence, the growth rate of western bay mussels declines, and less inorganic nitrogen is consumed by the (smaller) phytoplankton population. The nitrogen which is no longer utilised in the western bay issubsequently transported to the eastern bay, where it fuels greater phytoplankton production. Up to a stocking rate of ~10000 tonnes, this elevated production is able to offset the depletion due to increased mussel stocking, so that mussel growth rate remains approximately constant. This simulation provides an elegant demonstration of the subtle interactions which must be addressed when considering concurrent farming development in neighbouring areas.
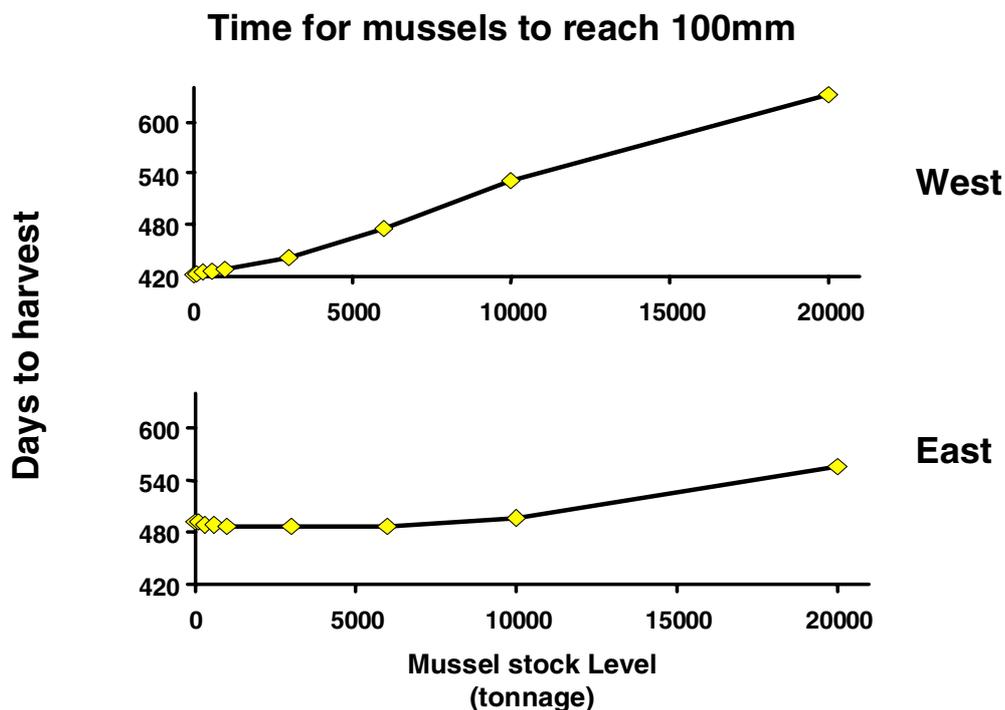
Quantifying Ecological Carrying Capacity

Ecological Carrying capacity is described as that level of farm development beyond which ecological impacts become unacceptable. This demands definition of ecologically acceptable impacts. Unfortunately, there are no agreed standards by which to define ecological acceptability.

Hatton *et al.* (2002) advocate the 'Limits of Acceptable Change' (LAC) approach to management of aquaculture development within the Firth of Thames. This approach accepts that it is difficult to define ecological acceptability, and advocates developing flexible criteria on the basis of current knowledge – with an acceptance that these criteria may need to be changed in the light of further knowledge. Nonetheless, it still demands that a suite of characteristics, and associated 'trigger-levels' (defined degrees of change) be agreed. If progress is to be made with adopting the LAC approach (or any other approach invoking Ecological Acceptability) within the Firth of Thames, then a substantial amount of work is likely to be required in order to define these characteristics, and the associated trigger levels.

Given that many of the possible far-field effects derive from the fact that shellfish may remove planktonic organisms from the water, it is essential that the 'diet' (*sensu* Mussel 'Diet') of *Perna canaliculus* and *Crassostrea gigas* be defined. Having determined what types of organism are removed, the next stage is to determine what quantity may be removed without having an unacceptable effect. Some questions might be:

a. We predict (for example) that the eggs and larvae of broadcast spawning benthic organisms will be removed; but are the number of successful recruits to the benthic population limited by the numbers of larvae which reach settlement age, or by other factors (such as the availability of suitable settlement sites).

b. We predict (for example) that there will be a 20% reduction in the abundance of phytoplankton within a radius of (for example) 500 m of this farm. Will such a reduction have an unacceptable impact upon the growth of zooplankton within this radius, and at the larger, bay-scale will this reduce the zooplankton population size unacceptably?

There is strong evidence that egg and larval survival determines year-class strength in fish, but we know of no studies that have determined the extent to which benthic populations are recruitment limited. Thus, question (a) is likely to require new experimental work focussed particularly upon benthic organims. With respect to (b), we have already established (Section 6.3), that, for all plausible phytoplankton concentrations, wide-scale depletion is likely to lead to corresponding reductions in the zooplankton growth rate. There is already a substantial literature regarding the relationships between food abundance and the growth rate of numerous zooplankton species (Hansen et al. 1997, for reviews; see Peters & Downing 1984), but we know of little, or no similar data for the larvae of fish and benthic organisms.

Most of the candidate characteristics by which ecosystem capacity may be measured (Table 5) consider only one, or a small part of the ecosystem. Similarly, the depletion model, the biophysical model and the production capacity model all restrict themselves to a small part of the ecosystem (nutrients, phytoplankton and perhaps, zooplankton). A more holistic, *a-priori* assessment of the partitioning of resources in the Firth of Thames

ecosystem could be made using a steady-state, mass-balance trophic model of the ECOPATH type (Christensen et al. 2000). The underlying model assumes linearity and (in its simplist form) equilibrium and spatial homegenity (with the associated restrictions governing extrapolation) but this model does offer the possibility (though not the certainty) of determining whether changes at the base of the foodweb will cascade up to higher members of the foodweb (fish, birds etc.). At the very least, this approach provides a formalised method by which to integrate a wide variety of data (usually from many, disparate sources) and determine whether or not they are mutually consistent. If they are not, the implication is that one's understanding of the ecosystem is at fault: suggesting that further investigations are required before a robust assessment of sustainability could be derived.

The ECOPATH approach has been used to explore management options for suspended scallop culture in Tongoy Bay (Northern Chile) (Wolff 1994) and other fisheries (e.g. Bundy (2001)), and NIWA are presently applying the approach to the Southern Plateau area of New Zealand's Exclusive Economic Zone (Bradford-Grieve et al. in prep.). In summary, the method involves subdividing the ecosystem's foodweb into several, user-defined compartments (e.g. phytoplankton, bacteria (both water column and sediment), ciliates, heterotrophic flagellates, mesozooplankton, macrozooplankton, fish larvae, fish, sea birds, benthos, detritus (water column and sediment)). For each compartment, the user supplies estimates of average annual biomass, production/biomass, consumption/biomass, ecotrophic efficiency (the proportion of a compartment which is utilized within the ecosystem), diet composition (what other compartments are exploited) and exports from the system. The method then aims to determine: firstly, whether the various rate estimates (which will be derived from many sources) are consistent with one another. If the data are consistent with one another, the method yields an indication of how tightly different components of the system are linked together (i.e. how much 'slack' there is in the system), and the user is able to explore the knock-on consequences of changing any component (in the context of this report: increasing the biomass of shellfish within the system). It is important to realise that, in its simplest form, the system does not aim to simulate detailed spatio-temporal dynamics (though it can be extended to do so), and assumes long-term mass balance. Unfortunately, though much of the requisite data are available for the Hauraki Gulf (and, to a lesser extent, the Firth of Thames), an analysis of this type is beyond the scope of this report.

The results of the Ecopath exercise would provide information relevant to both the issues of **Production capacity** and **Ecosystem capacity**.

8 Summary and Conclusions

The northern Firth of Thames sustains higher phytoplankton concentrations than do the Marlborough Sounds. For this reason, mussel growth rates are likely to be higher in the Firth of Thames. There are, however, substantial inter-annual variations in phytoplankton abundance and these will influence mussel growth.

Using a very simplistic model of phytoplankton utilization within mussel farms, it is predicted that a 50 ha farm will consume at least 25 % of the phytoplankton which pass through it – even in the northern Firth of Thames, where currents are largest. Comparison of the rates at which material is transported away from release sites in the northern Firth of Thames with generation-times for phytoplankton, microzooplankton, meso-zooplankton and benthic organisms / fish suggest that: **provided environmental conditions are favourable to their growth/regeneration** the impact of a farm is unlikely to extend more than 100s to 1000s of meters from a farm for nutrients, phytoplankton and microzooplankton. In contrast, the farm's influence may extend throughout the Firth for slower-growing organisms. In this latter case, the overall scale of aquaculture development will determine how much pristine water is available to dilute the incremental impact of each farm. Hence, it will determine whether far-field effects are detectable.

The Firth of Thames and Hauraki Gulf is a net consumer of organic material (animals consume more organic material than phytoplankton produce). Futhermore, average phytoplankton concentrations are probably insufficient to support maximal growth rates amongst the herbivorous filter-feeding community. These two observations may imply that even small reductions in phytoplankton abundance will greatly increase any food-limitation which other trophic groups suffer. Thus, there may be little scope to increase mussel stocks without adverse consequences.

Improved estimates of the Firth's production capacities could be arrived at by applying both the ECOSIM/ECOPATH approach and adapting NIWA's model of mussel production capacity in the Marlborough Sounds.

Improved estimates of the Firth's ecological capacity require: (a) experimental effort related to mussel filtering, (b) application of existing simulation models over a wider range of environmental conditions, (c) enhancements to existing simulation models.

₉ Appendices

Appendix 1. Interval Tests

Within the fields of environmental sciences and management (as in many others), the usual means of determining whether two data sets differ is through the use of one-, or two-sided tests of a point null hypothesis. McBride & Ellis (2001) make a persuasive argument that this is inappropriate. The fundamental reason is that, *a-priori*, we know that the null hypothesis which these tests seek to verify is genuinely false. Consider two series of random numbers drawn from the same distribution: it is vanishingly unlikely that any value (mean, standard deviation etc.) derived from these numbers will be **exactly** identical– yet this is the null hypothesis in traditional tests. A failure to detect a 'statistically significant' difference between corresponding characteristics derived from our two hypothetical series of random numbers merely indicates that our sample size is too small (ie, we have not drawn enough random numbers). Thus, whether or not a difference proves to be statistically significant is a function of sample size. Those who seek to detect a 'statistically significant difference' need merely to continue collecting data until they are able to demonstrate the difference (which, we know must exist). The second drawback of traditional tests is that they can only falsify the null hypothesis, they cannot **confirm** the null hypothesis.

In order to avoid the limitations associated with traditional tests of point hypotheses, (McBride & Ellis 2001) advocates the use of interval tests when addressing environmental issues. Tests of this sort are widely used in the pharmaceutical industry. Their great strength is that they do not force the user to test a null hypothesis which is known to be untrue ("**the true value of my chosen characteristic (of which the value derived from my sample data is an approximation) is exactly equal to this predefined value**"). Rather, the user tests the much more plausible null hypothesis: "**the true value of my chosen characteristic (of which the value derived from my sample data is an approximation) deviates from the predefined value by less than my chosen threshold level**". Furthermore, because interval tests are Bayesian, they require that the user specify a prior probability for the null hypothesis. The test yields a posterior probability – which may be larger (lending support to the null hypothesis) or smaller (perhaps sufficiently to favour the alternative hypothesis). Increasing the sample size will not inevitably lead to falsification of the null hypothesis. Rather, it will yield a more accurate prediction of the probability that this hypothesis is true. The null hypothesis may prove to be more likely than was originally thought!

Many people are averse to interval tests because they explicitly **require** the user to define the interval which they are testing (ie the range of values within which two corresponding characteristic will be considered 'not significantly different'). This is often

argued to be a subjective process– and it may well be! It is, however no more arbitrary than the traditional choice to adopt 5% as the acceptable probability of making a Type I error (rejecting the null hypothesis in error) when testing a point-value hypothesis. Indeed, we argue that the requirement which interval tests place upon the user to explicitly specify the thresholds beyond which differences will be considered 'significant' is a strength, rather than a weakness of the approach.   Specifically, they force the user to confront the natural variability inherent within biological systems and make a decision about what degree of change is **ecologically significant**.

We strongly advocate the use of interval tests as a means of determining whether the threshold, or trigger levels have been exceeded. We do so for two reasons: (a) they are based upon a realistic null hypothesis (and hence, the outcome of the test is less sample-size dependent), (b) because they explicitly enable the user to test hypotheses regarding **ecological significance**.

Appendix 2. Farm footprint analyses using biologiclly inert tracers.

The Numerical Model

Oldman & Senior (2000) employed an existing calibrated hydrodynamic model 3DD of the Greater Hauraki Gulf (Black et al. 2000) for their hydrodynamic simulations. This model was extensively tested against a large number of tide and current meter records throughout the Gulf and successfully simulates the circulation of waters within the Gulf. The horizontal grid resolution was 750m and 15 layers were used in the vertical.

To model the dispersal of a tracer from the release sites, the transport/dispersion model POL3DD (Black 1996) was used (see Oldman & Senior 2000) for details of the model and calibration parameters). POL3DD reads the hydrodynamic information from 3DD simulations and uses this information to calculate the spatial and temporal concentrations of tracer material as it disperses. The particle dispersal simulations presented in this report use the hydrodynamic simulations of Oldman & Senior (2000).

Tides, winds & stratification

A spring tide of amplitude 2.6 m was imposed at the open (seaward) boundary of the model. The time series of tide levels was extracted from the larger scale 750 m-grid model of (Black et al., 2000).

Oldman & Senior (2000) used wind speeds of 0, 8 and 15 m s^{-1} in their simulations, representing calm, average and the 95th percentile wind speeds for wind data recorded at the Mokohinau Islands (Outer Hauraki Gulf) between January 1998 and July 2000. Tracer dispersion modelling in this report is based on the hydrodynamic simulations using 15 m s^{-1} wind speeds, since the strong winds provide the largest particle dispersal. For this study we examine the effects of winds of 15 m s^{-1} from the NE, SE, SW and NW on the dispersal of a tracer from release sites in the Firth of Thames.

Oldman & Senior (2000) showed that the footprints of conservative-tracer released at Wilson Bay were similar in both well-mixed and stratified water bodies. The stratified simulations are used to drive the dispersal model for this study, using a typical spring temperature distribution. Stratification in early spring normally consists of warmer surface waters with underlying cooler water driven in from the outer Gulf (eg. Figure 3, of Oldman & Senior (2000)).

Appendix 3. The Depletion Model

The following describes a simple method for estimation of depletion rate in a proposed mussel farm site. Mussel depletion rate in a farm depends on structure of the farm, including length of longline, longline & crop longline densities, seed density, filtration rates of mussels, size composition of mussels and flushing time etc. Therefore, the estimation of depletion rate has to consider all related factors.

The method makes two important, but doubtful assumptions. Firstly, it assumes that the farm has no impact upon local current patterns. Secondly, each mussel within the farm is assumed to filter an independent volume of water. The first assumption may become invalid for very large farms. The second assumption is undoubtedly invalid (see main text for an explanation).

Definition of variables

❑ The farm
1. The area of a farm: A (ha)
2. Depth of the farm: D (m)
3. Volume of the farm: V (m^3)
4. Longline length: $Length_{Longline}$ (m)
5. Longline density (number of longlines in the farm): $D_{Longline}$ (ha^{-1})
6. Length of spat rope per longline: $Spat_{Longline}$ ($m\ m^{-1}$)
7. Length of crop rope perlongline: $Crop_{Longline}$ ($m\ m^{-1}$)
8. Spat density: D_{Spat} (no m^{-1})
9. The flushing time of the farm: $T_{Flushing}$ (h)
10. The refiltration ratio of all size class mussels: R_R (%)

❑ Mussels
The proposed farm is composed of spat and other size classes (n_{Size}) and the density of each class
1. Density of spat (e.g. 20-35 mm): D_{Spat} (no m^{-1})
2. Density of i size class: D_i= (no m^{-1}), where i refers i=1, 2, 3 ······
3. The number of size classes of crop: N, where N = i

❑ Clearance rates of individual animal of each size class
1. Clearence rate of spat (e.g. 20-35 mm): CR_{Spat}= ($m^3\ h^{-1}$)
2. Clearence rate of i size class: CR_i= ($m^3\ h^{-1}$)

Calculations:

1. Total number of longlines:

The number of total longlines depends on the size of a farm and longline density.

$$No_{Longline}=A \times D_{Longline} \text{ (no.)}$$

2. Number of longlines for spat and crop:

Calculation of number of longlines for each size class is based on assumption that 20% of the total longlines are for culture of spats and the rest are evenly divided for the culture of crops of other size classes. Therefore,

The number of spat longlines:

$$No_{SpatLongline}=No_{Longline} \times p \text{ (no.)}$$

The number of crop longlines:

$$No_{CropLongline}=No_{Longline} \times (1-p) \text{ (no.)}$$

Where p is the percentage of spat longlines in the farm. By default, it is set to 20%.

Therefore: the length of spat and crop longlines are calculated as:

The length of spat longlines:

$$Length_{Spat\_longline}=No_{SpatLongline} \times Length_{Longline} \text{ (m)}$$

The length of crop longlines:

$$Length_{Crop\_longline}=No_{CropLongline} \times Length_{Longline} \text{ (m)}$$

3. Length of culture rope for spat and crop:

Total length of spat and crop culture rope depends on the length of spat/crop longline and the length of spat/crop rope per longline.

Total length of spat rope:

$$Length_{Spat\_Rope}=Length_{Spat\_longline} \times Spat_{Longline} \text{ (m)}$$

Total length of crop rope:

$$Length_{Crop\_Rope}=Length_{Spat\_longline} \times Crop_{Longline} \text{ (m)}$$

4. Culture number of each size class

The culture number of each size class depends on the length of culture rope and density of that class. There are N size classes of crop and the rope length of each size class is assumed to be one N_{th} of the total length of crop rope (=$Length_{Crop\_Rope}$/N).

Total number of spat (eg. 20-35 mm)

$No_{Sapt}=Length_{Spat\_Rope} \times D_{Spat}$ (no.)

Total number of crop

$No_i=(Length_{Crop\_Rope} \div N) \times D_i$ (no.)

Filtration

Filtration of each size class is a result of clearance rate of individual, refiltration ratio and the number of mussels in that class. The sum of filtration of all size classes gives the total filtration.

Filtration of spats (eg. 20-35 mm)

$FR_{Spat}= CR_{Spat} \times No_{Spat} \div R_R$ (l)

Filtration of i size class

$FR_i= CR_i \times No_i / R_R$ (l)

By default, the length of mussels in size class of i=1, 2 & 3, are respectively, 35-60, 60-85, 85-110 mm.

Total filtration

$$FR_{Total}= FR_{Spat} \quad \sum_{i=1}^{N} FR_i \quad + \quad (m^3\ h^{-1})$$

Depletion rate (%)

Depletion rate is calculated from total filtration (FR_{Total}), flushing time ($T_{Flushing}$) and the volume of the farm (A).

Depletion (%) = $100 \times FR_{Total} \times T_{Flushing}$ / V

10 References

Anon (1992). The New Zealand Microbiological Water Quality Guidelines for

Recreational and Shellfish Gathering in New Zealand Waters 1992.

Bayne, B.L.; Thompson, R.J.; Widdows, J. (1976). Physiology: 1. **In**: Bayne, B.L. (ed) Marine Mussels: their ecology and physiology, pp. 506, Cambridge University Press, Cambridge.

Black, K.P. (1996). Lagrangian dispersal and sediment transport model POL3DD. Occasional Report **No. 21**, 69 p.

Black, K.P.; Bell, R.G.; Oldman, J.W.; Carter, C.S.; Hume, T.M. (2000). Features of 3-dimensional barotrophic and baroclinic circulation in the Hauraki Gulf, New Zealand. **New Zealand Journal of Marine and Freshwater Research 34**: 1-28.

Bradford-Grieve, J.M.; Probert, P.K.; Baker, A.N.; Best, H.A.; Boyd, P.; Broekhuizen, N.; Childerhouse, S.; Clark, M.; Hadfield, M.; Hall, J.A.; Hanchet, S.; Nodder, S.D.; Safi, K.; Thompson, D.; Wilkinson, I.; Zeldis, J. (in prep.). Pilot trophic model for subantarctic water over the Southern Plateau, New Zealand: a low biomass, high efficiency system.

Bradley, B.; Terralink NZ Ltd. (1999). The Penguin New Zealand Atlas. First. Penguin Books, Auckland. 199 p.

Brock, R.L.; Galbraith, G.R.; Benseman, R.A. (1985). Relationships of rainfall, river flow, and salanity to faecal coliform levels in a mussell fishery. **New Zealand Journal of Marine and Freshwater Research 19**: 485-494.

Broekhuizen, N. (1999). Simulating motile algae using a mixed Eulerian-Lagrangian approach: does motility promote dinoflagellate persistence or co-existence with diatoms? **Journal of Plankton Research 21(7)**: 1191-1216.

Broekhuizen, N. (2000). What fuels algal production in the Hauraki Gulf? **Water & Atmosphere 8(1)**: 10-12.

Bundy, A. (2001). Fishing on ecosystems: the interplay of fishing and predation in Newfoundland-Labrador. **Canadian Journal of Fisheries and Aquatic Science 58**: 1153-1167.

Carter, L.; Eade, J.V. (1980). Hauraki sediments. New Zealand Oceanographic Institute Chart, Coastal Series, 1:200,000.

Cembella, A.D.; Bauder, A.; Lewis, N.; Lawrence, J.; Manuel, J.; Lobsiger, U. (1997). "Population dynamics and spatial distribution of phycotoxic microalgae associated with shellfish aquaculture sites in Nova Scotia." Presented at the Oceans 1997, 557. 1.

Chang, F.H.; MacKenzie, L.; Till, D.; Hannah, D.; Rhodes, L. (1995). The first toxic shellfish outbreaks and the associated phytoplankton blooms in early 1993 in New Zealand. **In**: Lassus, P.; Arul, G.; Erad, E.; Gentien, P.; Marcallou, C. (eds). Harmful marine algal blooms, pp. 145-150, Lavoisier Intercept Ltd., Paris.

Chang, F.H.; Sharples, J.; Grieve, J.M.; Miles, M.; Till, D.G. (1998). Distribution of **Gymnodinium** cf. **breve** and shellfish toxicity from 1993 to 1995 in Hauraki Gulf, New Zealand. **In**: Reguera, B.; Blanco, J.; Fernandez, M.L.; Wyatt, T. (eds). Harmful Algae, pp. 139-145, Intergovernmental Oceanographic Commision of UNESCO, Eunta de Galicia.

Chang, F.H.; Zeldis, J.; Gall, M.; Hall, J.H. (in review). Seasonal and spatial variation of phytoplankton functional groups on the northeastern New Zealand continental shelf and in Hauraki Gulf. **Journal of Plankton Research**.

Christensen, V.; Walters, C.J.; Pauly, D. (2000). ECOPATH with ECOSIM: a user's guide. Fisheries Centre, University of British Columbia, Vancouver, Canada and ICLARM, Penang, Malaysia. [draft available at www.ecopath.org], Vancouver. 130 p.

Cole, R.G.; Grange, K.R. (1996). Under the mussel farm. **Seafood New Zealand (November)**: 25-26.

Crossland, J. (1980). The number of snapper *Chrysophrys auratus* (Forster) in the Hauraki Gulf, New Zealand, based on egg surveys in 1974-75 and 1975-76. **Fisheries Research Bulletin 22**: 1-38.

Crossland, J. (1981). Fish eggs and larvae of the Hauraki Gulf, New Zealand. **Fisheries Research Bulletin 23**: 1-61.

Cummings, V.J.; Thrush, S.F.; Hewitt, J.E.; Funnell, G.A. (2001). Variable effect of a large suspension-feeding bivalve on infauna: experimenting in a complex system. **Marine Ecology Progress Series 209**: 159-175.

Dahlbäck, B.; Gunnarsson, L.Å.H. (1981). Sedimentation and sulphate reduction under mussel culture. **Marine Biology 63**: 269-275.

Davenport, J.; Smith, R.J.J.; Packer, M. (2000). Mussels *Mytilus edulis*: significant consumers and destroyers of mesozooplankton. **Marine Ecology Progress Series 198**: 131-137.

Davies-Colley, R.J.; Bell, R.G.; Donnison, A.M. (1994). Sunlight inactivation of enterococci and fecal coliforms in sewage effluent diluted in seawater. **Applied and Environmental Microbiology 60(6)**: 2049-2058.

EPA. (1985). Chapter 6. Algae. **In**: Rates, constants, and formulations in surface water quality modelling., United States Environmental Protection Agency, Athens.

Eppley, R.W.; Rogers, J.N.; McCarthy, J.J. (1970). Half-saturation constants for uptake of nitrate and ammonium by marine phytoplankton. **Limnology & Oceanography 14**: 912-920.

Francis, M.P. (1993). Does water temperature determine year class strength in New Zealand snapper (Pagrus auratus, Sparidae)? **Fisheries Oceanography 2(2)**: 65-72.

Frost, B.W. (1975). A threshold feeding behaviour in *Calanus pacificus*? **Limnology & Oceanography 20(2)**: 263-266.

Gibbs, M.M.; Pickmere, S.E.; Woods, P.H.; Payne, G.W.; James, M.R.; Hickman, R.W.; Illingworth, J. (1992). Nutrient and chlorophyll *a* variability at six stations associated with mussel farming in Pelorus Sound, 1984-1985. **New Zealand Journal of Marine and Freshwater Research 26**: 197-211.

Giles, H. (2001). Modelling Denitrification in Continental Shelf Sediments. M.Sc. Thesis. University of Waikato. 110 p.

Grange, K.; Cole, R. (1997). Mussel farming impacts. **Aquaculture Update 19**: 1-3.

Greenway, J.P.C. (1969). Surveys of mussels (Mollusca: Lamellibranchia) in the Firth of Thames, 1961-67. **New Zealand Journal of Marine and Freshwater Research 3(2)**: 304-317.

Grenz, C.; Hermin, M.; Baudinet, D.; Daumas, R. (1990). In-situ biochemical and bacterial variation of sediments enriched with mussel deposits. **Hydrobiologia 207**: 153-160.

Grieg, M.J.; Proctor, R. (1998). A numerical circulation model of the Hauraki Gulf, New Zealand. **New Zealand Journal of Marine and Freshwater Research 22**: 379-390.

Hall, J.A.; Safi, K.A.; James, M.R. (in prep.). Microbial Dynamics in the Hauraki Gulf from spring to summer.

Hansen, P.J.; Bjørnsen, P.K.; Hansen, B.W. (1997). Zooplankton grazing and growth: scaling within the 2-2000 μm body size range. **Limnology and Oceanography 42(4)**: 687-704.

Hardin, G. (1968). The Tragedy of the Commons. **Science 162**: 1243-1248.

Hatton, S.; Inglis, G.; Thrush, S.; Zeldis, J. (2002). Marine Farming in the Firth of Thames - monitoring and performance criteria. NIWA Client Report **No. EVW02238**, 25 p.

Hawkins, A.J.S.; James, M.R.; Hickman, R.W.; Hatton, S.; Weatherhead, M. (1999). Modelling of suspension-feeding and growth in the green-lipped mussel *Perna canaliculus* exposed to natural and experimental variations of seston availability in the Marlborough Sounds, New Zealand. **Marine Ecology Progress Series 191**: 217-232.

Horsted, S.J.; Nielseon, B.G.; Rieman, B.; Pock-Steen, J.; Bjornsen, P.K. (1988). Regulation of zooplankton by suspension-feeding bivalves and fish in estuarine enclosures. **Marine Ecology Progress Series 48**: 217-224.

Ince, L.S.; Lutz, R.A. (1981). Modeling carrying capacities for bivalve molluscs in open, suspended-culture systems. **Journal of the world mariculture society 12**: 143-155.

Inglis, G.J.; Hayden, B.J.; Ross, A.H. (2000). An overview of factors affecting the carrying capacity of coastal embayments for mussel culture. Report for Ministry for the Environment **No. MFE00505**

James, M.R.; Weatherhead, M.A.; Ross, A.H. (2001). Size-specific clearance, excretion, and respiration rates, and phytoplankton selectivity for the mussel *Perna canaliculus* at low levels of antural food. **New Zealand Journal of Marine and Freshwater Research 35(1)**: 73-86.

Jillet, J.B. (1971). Zooplankton and hydrology of the Hauraki Gulf, New Zealand. New Zealand Oceanographic Institute Memoir **No. 53**, 103 p.

Kaspar, H.F.; Gillespie, P.A.; Boyer, I.C.; MacKenzie, A.L. (1985). Effects of mussel aquaculture on the nitrogen cycle and benthic communities in Kenperu Sound, Marlborough Sounds, New Zealand. **Marine Biology 85(2)**: 127-136.

Kendrick, T.H.; Francis, M.P. (in review). Fish assemblages in the Hauraki Gulf, New Zealand. **New Zealand Journal of Marine and Freshwater Research**.

Kleppel, G.S. (1993). On the diets of calanoid copepods. **Marine Ecology Progress Series 99**: 183-195.

Maddock, L.; Taylor, F.J. (1984). The application of multivariate statistical methods to phytoplankton counts from the Hauraki Gulf, New Zealand. **New Zealand Journal of Marine and Freshwater Research 18**: 309-322.

Mann, K.H.; Lazier, J.R.N. (1991). Dynamics of Marine Ecosystems. Blackwell Scientific Publications, Boston. 466 p.

McBride, G.B.; Ellis, J.C. (2001). Confidence of compliance: a Bayesian approach for percentile standards. **Water Resources 35(5)**: 1117-1124.

Morrisey, D.J.; Swales, A. (1996). Towards the development of guidelines for the sustainable management of aquaculture in the Auckland Region. NIWA Client Report **No. ARC304/1**, 170 p.

Nodder, S.N. (2000). Hauraki Benthics. Voyage Report **No. TAN9915**, 7 p.

Ogilvie, S.C.; Ross, A.H.; Schiel, D.R. (2000). Phytoplankton biomass associated with mussel farms in Beatrix Bay, New Zealand. **Aquaculture 181**: 71-80.

Oldman, J.W.; Senior, A. (2000). Wilson Bay marine farm dispersal modelling. NIWA Client Report **No. EVW01218/1**, 45 p.

Paffenhöfer, G.-A.; Orcutt, J.D. (1986). Feeding, growth and food conversion of the marine cladoceran *Penilia avirostris*. **Journal of Plankton Research 8(4)**: 741-754.

Peters, R.H.; Downing, J.A. (1984). Empirical analysis of zooplankton filtering and feeding rates. **Limnology & Oceanography 29(4)**: 763-784.

Proctor, R.; Greig, M.J.N. (1989). A numerical model investigation of the residual circulation in Hauraki Gulf, New Zealand. **New Zealand Journal of Marine and Freshwater Research 23**: 421-422.

Redfield, A.C. (1934). On the proportions of organic derivatives in sea water and their relation to the composition of plankton. **In**: James Johnstone Memorial Volume, Liverpool University Press, Liverpool.

Reeve, M.R.; Walter, M.A. (1977). Observations on the existence of lower threshold and upper critical food concentrations for the copepod *Acartia tonsa* Dana. **Journal of Experimental Marine Biology and Ecology 29**: 211-213.

Riemann, B.; Nielsen, T.G.; Horsted, S.J.; Bjørnsen, P.K.; Pock-Steen, J. (1988). Regulation of phytoplankton biomass in estuarine enclosures. **Marine Ecology Progress Series 48**: 205-215.

Rodhouse, P.G.; Roden, C.M. (1987). Carbon budget for a coastal inlet in relation to intensive cultivation of suspension-feeding bivalve molluscs. **Marine Ecology Progress Series 36**: 225-236.

Ross, A.H. (2002). Statement of Evidence before the Court. Evidence presented to Hawke Bay Regional Council concerning the application for a marine farm consent by Napier Mussel Ltd., 18 p.

Ross, A.H.; Gall, M.P.; Osborne, T.A.; Gibbs, M.M. (1998a). Does stratification control changes in phytoplankton abundance in New Zealand coastal inlets? **Seafood New Zealand 6(10)**: 32-37.

Ross, A.H.; Gibbs, M.M.; James, M.R.; Osborne, T.A. (1998b). Changes in nutrients affecting phytoplankton abundance in Pelorus Sound 1994-1998. **Seafood New Zealand 6(10)**: 32-37.

Scott, S.; Zeldis, J.; Pankhurst, N. (1993). Evidence of daily spawning in natural populations of the New Zealand snapper *Pagrus auratus* (Sparidae). **Environmental Biology of Fishes 36**: 149-156.

Sharples, J. (1997). Cross-shelf intrusion of subtropical water into the coastal zone of northeast New Zealand. **Continental Shelf Research 17**: 835-857.

Sharples, J.; Greig, M.J.N. (1998). Tidal currents, mean flows and upwelling on the northeast shelf of New Zealand. **New Zealand Journal of Marine and Freshwater Research 32**: 215-231.

Sinner, J.; Forrest, B.; Taylor, M.; Dodgshun, T.; Brown, S.; Gibbs, W. (2000). Options for a national pest management strategy for the Asian kelp *Undaria*. Technical Report to the Ministry of Fisheries (http://www.fish.govt.nz/information/undaria/tech-report.pdf), 63 p.

Sinton, L.W.; Davies-Colley, R.J.; Bell, R.G. (1994). Inactivation of enterococci and fecal coliforms from sewage and meatworks effluents in seawater chambers. **Applied and Environmental Microbiology 60(6)**: 2040-2048.

Sinton, L.W.; Hall, C.H.; Lynch, P.A.; Davies-Colley, R.J. (in press). Sunlight inativation of fecal indicator bacteria and bacteriophages from waste stabilization pond effluent in fresh and saline waters. **Applied and Environmental Microbiology**.

Stevens, C.; Spigel, B. (2001). Hydrodynamics of mussel farms. **Water & Atmosphere 9(4)**: 5.

Tenore, K.R.; Boyer, L.F.; Cal, R.M.; Corral, J.; Garcia-Fernandez, C.; Gonzalez, N.; Gonzalez-Gurriaran, E.; Hanson, R.B.; Isglesias, J.; Krom, M.; Lopez-Jamar, E.; McClain, J.; Pamatmat, M.M.; Perez, M.M.; Rhoades, D.H.; de Santiago, G.; Tietjen, J.; Westrich, J.; Windom, H.L. (1982). Coastal upwelling in the Rias Bajas, NW Spain: contrasting the benthic regimes of the Rias de Arosa and de Muros. **Journal of Marine Research 40(3)**: 701-772.

Tenore, K.R.; Corral, J.; Gonzalez, N.; Lopez-Jamar, E. (1985). "Effects of intense mussel culture on food chanin patterns and production in coastal Galicia, NW Spain." Presented at the International Symposium on the utilization of coastal ecosystems: planning, pollution and productivity, Rio Grande, Brazil, 321-328.

Wolff, M. (1994). A trophic model for Tongoy Bay - a system exposed to suspended scallop culture (Northern Chile). **Journal of Experimental Marine Biology and Ecology 182**: 149-168.

Zeldis, J.; Oldman, J.; Richards, J.; Ballara, S. (in review). Effects of cross-shelf transfer on zooplankton and larval fish survival in the Hauraki Gulf. **Marine Ecology Progress Series**.

Zeldis, J.R. (1992). Ichthyoplankton studies for fisheries research. Larval Fish Workshop, Australian Society for Fish Biology, Hobart 20 August 1991. **Bureau of Rural Resources Proceedings 15**: 19-26.

Zeldis, J.R.; Davis, C.S.; James, M.; Booth, W.; Ballara, S.; Chang, H. (1995). Salp grazing: effects on phytoplankton abundance, vertical distribution and species composition in a coastal habitat. **Marine Ecology Progress Series 126**: 267-283.

Zeldis, J.R.; Francis, R.I.C.C. (1998). A daily egg production method estimate of snapper biomass in the Hauraki Gulf, New Zealand. **ICES Journal of Marine Science 55(3)**: 522-534.

Zeldis, J.R.; Gall, M.; Uddstrom, M.; Grieg, M.J. (2000). La Niña shuts down upwelling in Northeastern New Zealand. **Water & Atmosphere 8(2)**: 15-18.

Zeldis, J.R.; Grieg, M.J.N.; Walters, R.A.; Image, K. (in prep.). Cross-shelf transfer, and new and remineralised nutrients on the northeast New Zealand Continental Shelf. **Continental Shelf Research**.

Zeldis, J.R.; Handley, S.; Gall, M.; Greig, M.; Pinkerton, M.; Richardson, K.; Hooker, S.; Moana, D.; Siddal, V. (2001a). From ENSO to Oysters. **Seafood New Zealand 9(11)**: 17-19.

Zeldis, J.R.; Richards, L. (in prep). Seasonal and spatial variation of mesozooplankton functional groups on the northeastern New Zealand continental margin.

Zeldis, J.R.; Sharples, J.; Uddstrom, M.; Pickmere, S. (1998). Fertilising the continental shelf: Biological oceanographic studies on the northeastern New Zealand continental margin. **Water & Atmosphere 6(1)**: 13-16.

Zeldis, J.R.; Smith, S.V. (1999). Water, salt and nutrient budgets for Hauraki Gulf, New Zealand. **In**: Smith, S.V. (ed) Australasian Estuarine Systems: carbon, nitrogen and phosphorus fluxes. **LOICZ Reports & Studies** No. 12, pp. 182,

Zeldis, J.R.; Thrush, S.; Oldman, J.; James, M.R.; Hooker, S. (2001b). Environmental monitoring program for Wilsons Bay Mussel Farm Developments. NIWA Client Report **No. EVW01218/1**, 21 p.