

# Review of the Ecological Effects of Farming Shellfish and Other Non-finfish Species in New Zealand

Cawthron Report No. 1476

August 2009

# Sustainable Aquaculture in New Zealand: Review of the Ecological Effects of Farming Shellfish and Other Non-finfish Species

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Prepared for

Ministry of Fisheries

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Recommended citation:

Keeley N, Forrest B, Hopkins G, Gillespie P, Clement D, Webb S, Knight B, Gardner J 2009. Sustainable Aquaculture in New Zealand: Review of the Ecological Effects of Farming Shellfish and Other Non-finfish Species. Prepared for the Ministry of Fisheries. Cawthron Report No. 1476. 150 p. plus appendices.

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# **EXECUTIVE SUMMARY**

#### Purpose and scope

Regional councils and the marine farming industry have identified a lack of publicly available information summarising the ecological effects of non-finfish marine farming in New Zealand (*e.g.* culture of mussels and oysters) as a critical information gap in relation to regional aquaculture planning and development. Although a large amount of research and monitoring has been conducted in New Zealand (especially with regards to Greenshell<sup>TM</sup> mussels), the information is generally not readily available to the public (*e.g.* Fisheries Resources Impact Assessment (FRIA) documents), or in a form that is easily assimilated by non-scientists. Hence, the Ministry of Fisheries (MFish) contracted Cawthron Institute (Cawthron) to conduct a review of existing information and to summarise knowledge of ecological effects associated with non-finfish species.

The scope and structure of this review reflects three specific objectives as prescribed by MFish; these are to collate and review information on the ecological effects of farming (1) Greenshell<sup>TM</sup> mussels including spat catching, (2) intertidal oysters, and (3) other non-finfish species. The other species considered include: dredge oysters, scallops, blue mussels, black-foot paua, sponges, crayfish, kina, sea cucumber and seaweeds. Greenshell<sup>TM</sup> mussel farming is, by far, New Zealand's largest aquaculture activity (~85% by area) and information pertaining to environmental effects is accordingly extensive. Oysters comprise a much smaller industry and locally derived literature is less common. The ecological effects of new and developing culture species are often unknown; in which case the guidance provided in this report is general in nature and based more on overseas examples and comparable species.

This report focuses on ecological effects only, whereas a broader range of effects and benefits on the coastal environment and communities will be relevant to resource management decisions (*e.g.* social and cultural aspects). The scope is limited to sea-based aquaculture and does not include managed wild fisheries like scallop enhancement. The review also focuses on coastal issues relating to farm structures and operations (including sea-rearing stages and spat catching) and does not consider wider 'off-site' effects (*i.e.* land-based hatchery rearing and product processing).

#### **Overview of effects**

The ecological issues of non-finfish farming are broadly classified into three main areas, including (1) those associated with effects on the seabed, (2) those associated with effects on the water column, and (3) wider ecological issues such as effects on fish, mammals and the spread of invasive species or disease (see figure over page). The typical effects associated with each of these categories are summarised in very general terms below. We encourage the reader to visit the relevant section of the report to obtain more detailed descriptions.

Through this review we identify that the environmental effects that arise from aquaculture are largely common to organisms that share cultivation methods (*e.g.* backbone suspended culture) and/or feeding strategies (*e.g.* filter-feeding bivalves). This is because most of the effects stem from either feeding and waste products or the physical presence of the structures themselves. In particular, our review highlights several commonalities among ecological effects arising from the subtidal cultivation of



Greenshell<sup>TM</sup> mussels and intertidal cultivation of Pacific oysters, particularly with regard to the seabed and water column. Moreover, farming of other filter-feeding bivalve species (*i.e.* scallops, flat oysters, blue mussels and subtidal Pacific oyster culture) are expected to have analogous effects on the marine environment, whereas the cultivation of organisms that require the addition of feed (*e.g.* paua, crayfish) may lead to different types or magnitudes of effects. The nature and magnitude of wider ecological effects such as the spread of pest species, disease outbreaks, or effects on the genetic makeup of natural populations will often depend on the species.



#### Seabed effects

The main ecological effects on the seabed from farming mussels, oysters and other filter-feeding bivalves arise from biodeposits and drop-off of shell and associated biota. In most instances, the severity of seabed effects has been assessed as low to moderate. The effects exhibit as minor enrichment of the seabed sediments (organic content increases by  $\sim$ 7.5%), increased build up of shell litter directly beneath the site, and in some instances increased aggregations of starfish and other epifauna taxa. Sediment enrichment, in-turn, affects the composition of sediment dwelling biota with productivity generally enhanced (*i.e.* some smaller species become more prolific). Changes to the surface dwelling biota (*e.g.* starfish) have been documented but are difficult to quantify and vary significantly between sites. Seabed effects are most pronounced directly beneath farm sites, reduce rapidly with distance, and are usually difficult to detect within 20-50 m away. The most important factors influencing the magnitude of effects are minimised by locating farms in well-flushed areas, where species and habitats of special value are not present.

Seabed effects from intertidal oyster farms are comparable to those from mussel farms, with the exception that there is increased scope for topographical changes to the seabed due to the positioning



of structures within the shallow intertidal zone. The potential for species other than mussels and oysters to affect the seabed has not been well studied, but we recognise that paua, crayfish and kina farming all require external feed inputs and will therefore have greater biodeposition and enrichment potential. At the other end of the spectrum, the seabed effects of algal culture are expected to be relatively benign, but there is potential for shading, or light limitation effects on the seabed.

#### Water column effects

Effects of mussel cultivation on the water column are less well defined than for the seabed, because water column characteristics are more dynamic and inherently harder to quantify. The physical presence of farms can alter and reduce current speeds, which affects water residence times and has implications for associated biological processes. Farm structures can also attenuate short-period waves, which can affect inshore ecology, but these issues are not considered significant at the present scale of development in New Zealand. Bivalves and other associated fauna release dissolved nitrogen (*e.g.* ammonium) directly into the water column, which can cause localised enrichment and stimulate phytoplankton growth. Toxic microalga blooms may lead to ecological or health problems, but there is no evidence of this being exacerbated by mussel farming in New Zealand waters. Filtration pressure by mussels is sufficient to potentially alter the composition of the phytoplankton and zooplankton/mesoplankton communities through feeding, but the extent to which this occurs and its ecological consequences are poorly understood. Despite the recognised knowledge gaps, the fact that no significant water column related issues have been documented suggests that effects associated with traditional inshore farming practices are minor.

Oysters and other bivalve species interact with the water column in a similar manner to mussels and hence the scope for nutrient enrichment, seston alteration and toxic microalgae bloom potential remains roughly equivalent. Differences in phytoplankton depletion potential can be estimated from equivalent farming densities (per  $m^2$ ) and filtration rates. However, intertidal culture of oysters is thought to have slightly more profound effects on hydrodynamics due to the structures occupying a cross section of the water column. Species that require atypical cage structures or additional feed (*e.g.* crayfish and paua) will likely interact with the water differently; the extent to which their effects will differ remains undetermined.

#### **Carrying capacity**

There are no definitive studies that provide a clear answer to the question of carrying capacity in relation to New Zealand mussel farming. Spatial modelling tools offer a way of estimating the extent to which the cumulative effects of mussel farming may be approaching ecological carrying capacity on "bay-wide" and "regional" scales. However, knowledge gaps are still evident in these models; particularly in the biological aspects, which are still areas of active research. Production is closely linked to Southern Oscillation Index (SOI) and can be projected from climate data; before this relationship was understood there were concerns that carrying capacity had been reached within the Marlborough Sounds system. However, mussel growth and production have subsequently recovered, suggesting that farming levels were not the driving factor. The present farming intensity in New Zealand is still considered low-moderate in an international context. There are anecdotal reports that Pacific oyster production has approached carrying capacity in some New Zealand estuaries, although this has not been conclusively proven. The potential for such effects is situation-specific and temporally variable.

#### Wider ecological issues

The wider ecological issues that are assessed include: habitat creation and alteration, effects on fish, seabirds and mammals, biosecurity risks, disease and genetics. These issues are generally less well studied than seabed and water column effects, due either to logistical difficulties in obtaining quantitative data, lack of awareness, or because the need has not arisen (*i.e.* potential for adverse effects is generally perceived to be low).

Habitat creation recognises the fact that, in addition to growing the culture species, farms function as mid-water artificial reefs. Artificial structures provide novel foraging habitat, detrital food sources, breeding habitat, and refuge from predators for some species, and can contribute to seabed enrichment issues through biodeposits and drop off. As well as changing habitat characteristics, and with it, the composition of the wild fish assemblages, marine farms can affect fish populations through changing fishing pressures and aggregation behaviour. In addition, recruitment of fish larvae to wild fish populations could theoretically be affected by the filtration pressures of large bivalve farms. The role of aquaculture structures as reservoirs for the establishment of pest organisms (*e.g.* fouling pests) is also recognised. The development of aquaculture in New Zealand therefore has the potential to exacerbate the domestic spread of pest organisms, although various management approaches can be implemented to reduce such risks. Some of these approaches (*e.g.* codes of practice, treatments for infected seed stock) have already been implemented by aquaculture companies in New Zealand in response to existing pests.

Potential effects on seabirds and marine mammals (seals, dolphins and whales) relate mainly to habitat modification, entanglement in structures and habitat exclusion. For seabirds, a range of potential effects are recognised, but these are generally not well understood. The few overseas studies describing seabird interaction with oyster culture sites provide no evidence of adverse effects. Our review only revealed one reported case of marine mammal entanglement in mussel (spat) farm structures in an industry that now comprises ~900 farms, and there remains some uncertainty over whether the death was the result of entanglement. Hence, scope for marine mammal interactions and/or exclusion has been low, possibly due to the historical 'coastal ribbon' style development, but this may change with the advent of large offshore mussel farming sites. The potential for adverse interaction between intertidal oyster culture and marine mammals is minor in New Zealand, as there is probably minimal overlap between sites of intertidal cultivation and typical marine mammal habitat.

The risk of transmission of pathogens or parasites from cultured to wild mussels to other species is considered minimal at present. With the exception of one protozoan parasite, all other diseases reported in cultured mussels have been less prevalent than from wild mussels. No mussels in New Zealand have been reported with any recognised important pathogens. Likewise, there have been no documented serious (OIE listed) parasites/pathogens of Pacific oysters in New Zealand. But several other less severe diseases and parasites have been reported to occur. Most of these are also globally ubiquitous and may pose some commercial threat to oyster production, but not the wider ecology. Overall, disease has not been a significant issue with New Zealand aquaculture, but disease propensity is species-specific and therefore needs careful consideration with the introduction of other new culture species.



One of the less recognised effects of non-finfish aquaculture concerns maintenance of genetic diversity. Studies of genetic structuring within populations of *Perna canaliculus* have consistently demonstrated high levels of genetic variation, and more recently, introgression and viable hybridisation between northern and southern populations has been identified. However, the predisposition of Greenshell<sup>TM</sup> mussels to genetic issues is likely to be mitigated by high connectivity among mussel populations, and the industry being based on wild-sourced progeny. Furthermore there is already a high pre-existing level of inter-regional mussel seed-stock transfer. Therefore, the continued transfer of wild-sourced mussels within and between the northern and southern groups is unlikely to adversely affect fitness of wild stocks in the future. However, this conclusion does not hold if the mussel industry were to increase its dependence on hatchery-supplied spat. In the case of Pacific oyster cultivation, ecological effects on wild populations are not as relevant since Pacific oysters are non-indigenous to New Zealand. Furthermore, recent advances in breeding and the future production of triploid oyster spat that are sterile will likely eliminate the potential for genetic interactions.

#### Synthesis of findings

A case study of environmental risk associated with oyster farming was used to evaluate the relative significance of the different ecological effects. Overall, environmental risks to the seabed and water column were considered low, with the largest risks associated with moderate effects to the seabed in close proximity to the farms. With regard to the wider ecosystem effects, biosecurity issues relating to the spread of pest organisms received the highest risk ranking. The spread of pest organisms by aquaculture activities can occur at regional scales, potentially leading to ecologically significant and irreversible changes to coastal ecosystems. The potential for disease also scored relatively high; even though the likelihood was scored as relatively low, the overall score was inflated by the fact that the consequences of an outbreak could be significant, the effects long-term or irreversible and at greater than local scales. There are a range of remaining effects categories relating mainly to water column and wider ecosystem change for which ecological significance was on average scored as low or very low.

The results from the risk assessment of oyster farming are broadly transferable to the subtidal cultivation of Greenshell<sup>TM</sup> mussels and other bivalve species. Perhaps the main point of difference would be in relation to seabed effects and changes in topography, which are different for intertidal versus subtidal cultivation due in large part to the proximity of the structures to the seabed and the water depth. The magnitude, spatial extent and duration of effects arising from the spread of pest species and/or disease is considered high for all cultured non-finfish species; however, the pests and/or diseases involved are likely specific to the type of cultivation or species.

While the notion of ecological 'risk' tends to imply negative or adverse effects, there are some ecological effects from farming shellfish that could subjectively be considered 'neutral' or 'beneficial'. Furthermore, when the range of effects is considered as a whole it could be argued that some nominally 'adverse' effects may be compensated to some extent by more 'positive' effects. For example, although natural seabed sediments and benthos may be altered beneath oyster and mussel farms, local biodiversity and production may be enhanced through provision of habitat for fouling.



Hence, these types of considerations need to be part of management discussions in relation to aquaculture developments.

#### Future mitigation and management

Possible effects associated with likely future developments in the aquaculture industry such as the imminent move to larger offshore areas and the conceptual implementation of integrated (multi-species) culture are considered. These developments are considered to have some potential for managing and mitigating the effects that are conventionally associated with inshore, single species aquaculture. However, possible scale-related effects of offshore aquaculture remain undetermined and difficult to predict. Another key management and mitigation measure that is identified throughout this review, is careful site selection; *i.e.* sites that have strong currents to aid waste dissipation, good food supplies in the case of extractive culture (*i.e.* suspension feeding bivalves) and are removed from sensitive habitats. It is also recognised however, that in near-shore regions, areas that have strong current tend to coincide with ecologically significant or sensitive habitats, creating a trade-off between waste-dispersion potential and localised effects on such areas.

Effects can also be mitigated through adopting management strategies for pest species, disease and genetic diversity. Approaches to managing pest species are likely to be transferable across different forms of aquaculture. However, in the cases of disease and genetics, management will differ between species and more research may be required before appropriate protocols can be developed. This is especially the case with the minor and potential culture species in New Zealand, for which relatively little is known.

Overall, this review highlights that our present knowledge of ecological effects arising from New Zealand's two most widely practiced forms of aquaculture (Greenshell<sup>TM</sup> mussel and Pacific oyster farming) is reasonably good. This is especially true for the more recognised effects associated with the seabed, and, to a lesser degree, water column processes. It is also apparent that the level of knowledge is high when put in the context of effects associated with other coastal activities (*e.g.* fishing and trawling, habitat loss and non-point source pollution). However, through this review we have also identified areas where our knowledge is lacking or can be improved and these are listed toward the end of the document.

Even more broadly, we suggest that management responses to farm developments be made in relation to other sources of environmental risk to estuarine systems at a bay-wide or regional scale, so that the effects of aquaculture are placed in context. Such an approach can be applied in defined regions (*e.g.* estuaries) or across multiple regions, providing a robust basis for developing plans for research and prioritising management according to the greatest sources of risk.

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## GLOSSARY

Term/abbreviation	Definition
AMA	Designated aquaculture management area
Benthic	Bottom dwelling
Biodeposition	Depositional flux (downward 'rain') of biodeposits; in this context primarily
_	constitutes faeces and pseudofaeces from bivalves
Conspecifics	Of the same species
Epibiota	Plants and/or animals living on top of the seabed
Epifauna	Animals living on top of the seabed
FRIA	Fisheries resource impact assessment
HAB	Harmful algal bloom – blooms of phytoplankton that have potentially toxic
	properties (to other wildlife or humans when consumed).
Infauna	Small animals living within the sediment matrix
Introgression	In genetics, is the movement of a gene (gene flow) from one species into the
	gene pool of another by backcrossing an interspecific hybrid with one of its parents.
Macrobenthos	Benthic organisms that are retained by 500 µm mesh sieve, mainly arthropods
	(insects and amphipods), annelids (polychaetes, oligochaetes), and molluscs
	(bivalves, gastropods).
Microbenthos	Very small benthic organisms <i>e.g.</i> bacteria, fungi, protozoa, microscopic algae
MFA	Marine Farming Association
MSQP	Marlborough Shellfish Quality Program: an industry (MFA) funded food
-	safety program which tests shellfish and seawater for biotoxins (including
	phytoplankton composition), bacteria and heavy metals.
Phytoplankton	Microscopic marine algae living part or all of life in water column
Planktonic	Animals that live and drift within the water column
QMS	Quota management system
REDOX	Shorthand for reduction-oxidation reaction. Usually measured as REDOX
	potential on a scale that indicated reduction (addition of electrons) and
	oxidation (removal of electrons) for a given material. Expressed in millivolts.
Sedimentation	Natural process by which organic and/or inorganic particles settle out of the
	water column, to the seabed. Can be enhanced by farming due to production
	of (waste) biodeposits from shellfish or animal feeds.
Sessile	Attached to a substrate. Non-motile.
Seston	Drifting particles in the water column. Includes living plankton, detritus and
	sediment.
SOI	Southern oscillation index - calculated from the monthly or seasonal
	fluctuations in the air pressure difference between Tahiti and Darwin.
SPM	Suspended particulate matter (in water column – usually assessed
	gravimetrically)
TAC & TACC	Total allowable catch & total allowable commercial catch
Temporal	Variance with time
Zooplankton	Animal plankton, usually macroscopic and feed on algae



## 1. INTRODUCTION

#### 1.1. Purpose

The sustainable development of marine aquaculture in New Zealand needs to be based on a sound knowledge of the environmental effects in relation to the various culture species, farming methods and environments. A collation and review of knowledge relating to the ecological effects of finfish aquaculture was undertaken in 2007 (Forrest et al. 2007) under contract to the Ministry of Fisheries (MFish). Regional councils and the marine farming industry have identified a lack of publicly available information summarising the ecological effects of other types of marine farming in New Zealand (*i.e.* culture of non-finfish species) as a critical information gap in relation to regional aquaculture planning and development. Although a large amount of research and monitoring has been conducted in New Zealand on the environmental effects of some types of non-finfish aquaculture (especially Greenshell<sup>TM</sup> mussels), the information is generally not readily available to the public, or in a form that is easily assimilated by non-scientists. Among the related concerns are that misunderstanding of the effects of marine farming on the coastal marine environment, or gaps in existing knowledge, are likely to constrain good aquaculture decision-making. In order to overcome these constraints MFish contracted Cawthron Institute (Cawthron) to conduct a review of existing information to summarise and place into context knowledge of ecological effects ssociated with non-finfish species.

The overall objective of this report as specified in AQE2008/02 contract documents is to: Summarise and review existing information on the ecological effects of farming shellfish and other species (excluding finfish) in the New Zealand marine environment, using existing information in New Zealand and overseas examples where appropriate or necessary.

The outcomes of the review are expected to assist regional councils, communities and the marine farming industry plan for and develop sustainable aquaculture in New Zealand.

#### 1.2. Scope and report structure

The scope of this review, and structure of the report, largely reflects the following three specific objectives that were prescribed by MFish in the brief:

- 1. To collate and review information on the ecological effects of farming mussels (*Perna canaliculus*), including offshore mussel farming and spat catching, in the New Zealand marine environment.
- 2. To collate and review information on the ecological effects of farming Pacific oysters (*Crassostrea gigas*) in the New Zealand marine environment.
- 3. To collate and review information on the ecological effects of farming species other than mussels (*P. canaliculus*), oysters, and finfish, in the New Zealand marine environment.

In addition to excluding finfish species, the scope of this study is limited to sea-based aquaculture (also commonly referred to as 'mariculture'). Managed wild fisheries such as

enhancement of paua, scallop or cockle populations and land-based aquaculture are excluded. The main points of difference being firstly, that sea-based aquaculture interacts directly with the surrounding marine environment, whereas, land-based systems are funnelled to a single outlet, which facilitates diversion and treatment of effluent prior to being discharged back to the marine environment if necessary. Secondly, enhancement of wild fisheries involves grow-out of populations in their natural habitat, and usually on the seabed (and cannot be owned by individuals), which has a suite of potential ecological issues that are not necessarily relevant to aquaculture.

Sections 2 and 3 review information on the ecological effects of Greenshell<sup>TM</sup> mussel and Pacific oyster cultivation, respectively, reflecting the dominance of these sectors in New Zealand. These sections also provide a synopsis of the historical development of each sector, along with more recent developments such as moves for offshore cultivation of mussels. The review of mussel farming effects also includes additional quantitative meta-analyses of data produced as part of Fisheries Resources Impact Assessment (FRIA) studies. FRIAs are environmental investigations that were conducted on behalf of mussel farming companies or consortia by various research providers (primarily Cawthron and NIWA) as part of information requirements for MFish's permitting process for aquaculture development. The review of oyster farming effects in Section 3 considers only Pacific oyster cultivation, and follows a similar structure to that for mussels. The main difference is that there is scant information regarding the effects of Pacific oyster cultivation in New Zealand, hence we draw more heavily on knowledge from overseas where this is considered appropriate.

Section 4 considers the actual or potential ecological effects arising from the aquaculture of non-finfish species other than Greenshell<sup>TM</sup> mussels and Pacific oysters. This considers minor New Zealand species that are either already cultivated at a commercial scale, but in small quantities, or have significant potential and are presently undergoing growth trials or research. Hence, in this section we include dredge oysters (*Ostrea chilensis*), scallops (*Pecten novaezelandiae*), blue mussels (*Mytilus galloprovincialis*), black-foot paua (*Haliotis iris*), sponges (various), crayfish (*Jasus edwardsii*), kina (*Evechinus chloroticus*), sea cucumber (*Stichopus mollis*) and seaweeds (various). Because of the paucity of information for New Zealand, this section draws heavily on information from overseas and our knowledge of effects associated with aquaculture of other comparable species.

As such, this review is based on information from a variety of sources including international and national journals, 'grey' literature (*e.g.* consultancy reports), and personal communications with stakeholders. Within each main section of the report, we consider the full spectrum of potential aquaculture environments where applicable. For example, details relating to the various methodologies for mussel spat collection (for culture) are provided at the end of the mussel section (Section 2.6). However, issues associated with new and developing forms of aquaculture, such as offshore farming and integrated culture, are discussed later in Section 6. At the end of the report we assess the relative risk and significance of actual or potential ecological effects (Section 5), to identify any significant information gaps (Section 8) and provide guidance on the salient issues for future management (Section 7).



This report focuses on ecological effects only, whereas a broader range of effects and benefits on the coastal environment and communities will be relevant to resource management decisions. For example, future mussel farming development has been shown to be limited by the amount of farming activity that is socially acceptable (Banta & Gibbs 2007). As a result the social-carrying capacity for mussel farming within a given area may ultimately constrain production to levels that are sustainable with regard to ecological effects. 'Ecological effects' does however encompass ecological systems, processes and values that contribute towards an assessment of 'Natural Character', which is an important aspect of New Zealands' Resource Management Act (RMA). The information contained within may therefore be relevant to assessments of natural character under the RMA process.

It should also be recognised that we limit the scope of the report to a discussion of coastal issues relating to farm structures and operations (*i.e.* the sea-rearing stages of aquaculture and any associated spat catching). We do not consider the wider suite of 'off-site' effects that are generally recognised for aquaculture, such as those from land-based hatchery rearing and product processing (*e.g.* ICES 2005; McKindsey *et al.* 2006). Similarly, we do not consider any of the short-term effects that may arise from farm construction, as we assume these are minor by comparison with ongoing effects from aquaculture operations. Finally, the report presents information that is of a general nature only. It is not intended to be an assessment of environmental effects that could be used directly in relation to permit applications; any assessment for such purposes would need to consider a range of site-specific effects.

#### 1.2.1. Structure of sub-sections

Within each of Sections 2-4 we provide a review and synthesis of the pertinent literature, and discuss present and potential management and mitigation strategies as appropriate. The length of each section and sub-section in part reflects where the depth of knowledge is greatest rather than the actual or potential significance of each issue. Similarly the order in which we present the information is not meant to imply importance; it merely reflects the way we have chosen to structure the subject material to help the readability of the report, especially given that many of the sub-sections are inter-related. In order for the reader to discriminate between the New Zealand situation compared with overseas experience, we use paragraph divisions throughout each sub-section as appropriate. For some issues, however, New Zealand and overseas experience is similar, and the available knowledge is collectively discussed in the text to avoid repetition. Where this collective approach has been taken it is stated as such at the beginning of each relevant section. We have attempted to make each of the main subsections as 'stand-alone' as possible, at risk of creating repetition in parts of the overview and introductory sections.

#### 1.3. Background to New Zealand aquaculture industry

Marine aquaculture is a rapidly developing industry in New Zealand, based primarily on subtidal long-line cultivation of Greenshell<sup>TM</sup> mussels (*P. canaliculus*), intertidal rack

cultivation of Pacific oysters (*C. gigas*) and sea-cage farming of King salmon (*Oncorhynchus tshawytscha*). The broad geographic distribution of these industry sectors (Figure 1), and related production and market statistics (Table 1), illustrate the overall dominance of Greenshell<sup>TM</sup> mussel cultivation at a national scale, within a regional focus for Pacific oysters (primarily Northland and Auckland) and salmon (primarily the Marlborough Sounds). The combined market revenue from mussels, oysters and salmon exceeds \$350 million per annum, and approximately 7,767 ha of coastal water space is currently used or consented for aquaculture in New Zealand<sup>1</sup>.

Compared with oysters and salmon, the mussel industry has undergone a relatively rapid expansion in the past decade, which has included both intensification of farming in long-established growing regions and more recent moves to develop relatively large offshore blocks. This rapid phase of development in recent times is reflected in a 56% increase in coastal space allocated for aquaculture between 2001 and 2008 (source: www.aquaculture.govt.nz).

The goal of the New Zealand Aquaculture Strategy is that the aquaculture sector will have sales of \$1-billion per annum by 2025 (NZAS 2006), with growth expected to come from existing species, as well as from development of new species and products. Currently, seabased culture of species other than mussels, Pacific oysters and salmon is limited to a few small-scale abalone (most of which are land-based) and seaweed (*e.g. Macrocystis pyrifera* and *U. pinatifiida*) farms. There are also a number of species with recognised potential presently undergoing growth trials (*e.g.* flat oysters, scallops, kina, crayfish, groper and kingfish) or more preliminary investigative research (*e.g.* geoduck, clams and other fish species).

Table 1.	Aquaculture industry farm statistics from three primary species. Source: New Zealand
	Aquaculture Council Annual Report 2006-2007.

Species	Number	Total ha	Tonnes	Sales (NZ \$ Million)		
	of farms	marine space	harvested	Domestic	Export	Total
Greenshell <sup>TM</sup> mussels	645	4,747	97	43	181	224
King salmon	23 <sup>a</sup>	60	7.7	59	42	101
Pacific oysters	230	750	8.8	14	18	32

<sup>a</sup> Only eight salmon farms are currently in operation.

<sup>&</sup>lt;sup>1</sup> Source: New Zealand Aquaculture Council Annual Report 2006-2007 (does not include the scallop enhancement area or unoccupied space).





Figure 1. Geographic locations of main marine farming activities in New Zealand.

Table 2.List of current permitted aquaculture space for each main New Zealand region, indicating<br/>allocated AMA space and dominant form of aquaculture (Figure 1). Currently there is no marine<br/>farming in East Cape, Taranaki, Manawatu-Wanganui, Nelson and Otago regions. Data provided<br/>by respective Regional Councils and verified by MFish.

	Consented	
Region	area (ha)	Main activity
Tasman	6086	Mussel farming and mussel and scallop spat catching (and active
		enhancement of wild scallop fishery)
Bay of Plenty	3810	One new large offshore AMA yet to be developed, and some small
		oyster and mussel farms
Marlborough	3056	Mussel farming, some salmon farming and trials with new species
Hawke Bay	2469	One large offshore site currently undergoing trials with shellfish
Waikato	1044	Mostly mussel farms and some oyster farms
Northland	705	Mostly oyster farms and some mussel farms
Auckland	326	Mostly oyster farms and some mussel farms
Southland	286	Mostly mussel farming and some salmon farming
Canterbury	179	Several small mussel farms along with some salmon and paua
		farming. One new large (2695 ha) offshore AMA is still under
		consideration by MFish.
West Coast	46	One mussel farm in Jackson's Bay
Chatham Island	8	One disused site
Wellington	4.3	Few small trial sites



# 2. ECOLOGICAL EFFECTS OF GREENSHELL<sup>™</sup> MUSSEL FARMING

#### 2.1. Introduction

The first 'mussel rafts' in New Zealand appeared in the Marlborough Sounds in 1967 (Dawber 2004). Raft culture of Greenshell<sup>TM</sup> mussels expanded steadily until 1975, at which point long-line culture and some innovative processing methods revolutionised the industry, and stimulated a rapid expansion of mussel farm sites. The long-line culture method, on which the industry is now exclusively based, involves growing mussels on a continuous rope (long-line) that is suspended in the water column from a floating double stranded 'backbone' line that is anchored to the seabed at each end (Figure 2 and 3). The greatest intensity of mussel farming presently occurs in the sheltered waters of the Marlborough Sounds, with ~889 current consented Aquaculture Management Areas (AMAs) occupying ~3,061 ha. Other historically important mussel aquaculture areas include the Firth of Thames in the Waikato (and Auckland) Regions and to a lesser extent Bank Peninsula in Canterbury (Figure 1). The industry has traditionally been based on the coastal ribbon development comprising relatively small farms (~3-5 ha in area) situated close to shore.

In recent years, pressure for mussel farming space and changes to the legislation surrounding space allocation (*i.e.* Aquaculture Reform Act 2004) has lead to the creation of a number of comparatively large AMAs and an associated shift of long-line mussel culture from sheltered harbours to more exposed open coastal sites. The largest aquaculture region is presently Tasman, with 6,086 ha of permitted space<sup>2</sup>; 4010 ha of which is used solely for scallop spat catching, ~900 ha for mussel spat catching and the remainder for mussel grow-out. Much of the mussel farming space in Tasman has only recently been permitted and is yet to be fully developed. The Waikato region is presently New Zealand's largest, developed multi-ownership AMA, with 1220 ha of farmed space comprising >300 individual sites, primarily used for growing mussels<sup>3</sup>.

A single, large 2,465 ha AMA offshore from Hawke Bay is currently undergoing commercial trials and may be developed in the near future. This was the largest and first true 'open ocean' site to be consented in New Zealand intended primarily for farming mussels. Subsequently, a 3,800 ha site offshore from Opotiki two and a smaller, exposed site at Jackson Bay have been permitted, and a further 2,695 ha site offshore from Banks Peninsula (Pegasus Bay) is awaiting a final decision from MFish. Each of these new offshore sites represents a significant increase in available aquaculture space for the regions concerned and New Zealand as a whole. The status of offshore aquaculture and the associated environmental issues are discussed further in Section 6.1.1.

Until recently, seed-stock supply for the entire mussel industry was dependent on the availability of 'Kaitaia spat'. This term refers to spat that is attached to seaweed which is

<sup>&</sup>lt;sup>2</sup> As of September 2008

<sup>&</sup>lt;sup>3</sup> At the time of writing 520 ha of this space was still under consideration by MFish



naturally (*e.g.* after storms) deposited on the beach northwest of Kaitaia in Northland (Figure 1). At present, Kaitaia spat supplies ~80% of industry needs (G Coates, pers. comm.), although this figure varies greatly from year to year (Forrest & Blakemore 2002). Reliability issues with this natural source have meant that dedicated spat catching is increasingly undertaken in other regions. The Marine Farming Association presently operates five permanent spat catching and eight spat holding farms in the Marlborough Sounds plus two seasonal spat catching sites in Golden Bay and Tasman Bay. There is also a small scale spat catching operation in Aotea Harbour on the west coast of North Island. A more detailed description of spat catching methods can be found in Section 2.6.



**Figure 2.** Typical long-line mussel farming in the Marlborough Sounds: a,b. arrays of double backbone lines beneath which a continuous culture long-line is suspended; c,d. long-line cultures in the water column (Cawthron).

#### 2.1.1. Scope of this review and information sources

This review (Sections 2.3 to 2.5) focuses primarily on the grow-out stage of mussel cultivation. From the available information on spat-catching effects in New Zealand, it is apparent that there are few significant ecological issues that arise, and no issues that are likely to be of more significance than described for the cultivation phase. A summary of spat-catching effects is provided in Section 2.6 and draws largely on knowledge of effects from mussel cultivation. We note that Section 2.6 describes the site-specific effects of spat catching; there are a range of



issues that arise as a result of the national transfer of spat within and between catching and cultivation regions areas, which we address separately within Sections 2.3 to 2.5.

The considerable body of available information on mussel cultivation effects comes both from New Zealand and overseas. In New Zealand, much of this information is found in the grey literature, although published information is available for certain key ecological issues. We do not attempt to source or summarise all of the unpublished reports on mussel farming effects in New Zealand, rather we base our assessment on a representative cross-section of these reports and more recent work that synthesises the findings of prior studies. Much of the information that has been taken from specific client reports (*e.g.* FRIAs) is used to formulate a picture of 'typical' types and magnitudes of effects. Reference to specific sites is deliberately limited. Where appropriate and necessary, we also draw on overseas literature describing the ecological effects of cultivating different mussel species (primarily blue mussels *Mytilus* spp.). We make the reasonable assumption that effects described in overseas studies of different mussel species will be at least qualitatively similar to Greenshell<sup>TM</sup> mussel effects where suspended cultivation methods are used in comparable coastal environments.

A notable shortcoming of the published information is that knowledge of ecological effects is typically limited to only one or a few mussel farm sites considered in isolation, reflecting the level of effort required to obtain an in-depth understanding of effects and related processes (Kaspar *et al.* 1985; Chistensen *et al.* 2003; Hartstein & Rowden 2004). Hence, to supplement present knowledge, our review draws on results obtained from additional analyses of New Zealand data that were generated as part of industry-funded field studies conducted by Cawthron. These studies were undertaken to meet MFish FRIA information requirements under the marine farm permitting process. The FRIA analyses we present in this report are limited in scope in that they describe only key indicators of seabed or water column effects from mussel farms. However, this additional work is unique in that it represents the first comprehensive effort to characterise long-line mussel farm effects (for ~54 farm sites in the Marlborough Sounds) across a range of environments in relation to key environmental (*e.g.* water currents) and other (*e.g.* farm age) factors that could influence the nature and magnitude of effects.

#### 2.2. Overview of ecological issues

The ecological issues of mussel farming can be broadly classified into three main areas, including (1) those associated with effects on the seabed, (2) those associated with effects on the water column, and (3) wider ecological issues such as effects on fish, mammals and the spread of invasive species or disease (Figure 3). Efforts to document the ecological effects of mussel farms both overseas and in New Zealand have focused on the effects on seabed sediments and associated biota (see Section 2.3). The primary effects arise as a result of organic enrichment of the seabed by deposition of mussel faces and pseudofacees<sup>4</sup>, as well as

<sup>&</sup>lt;sup>4</sup> Pseudofaeces are mucous-bound aggregates of particles that are filtered but not consumed by mussels and other filter feeders.



drop off of mussels, shells and biofouling organisms. In New Zealand, a long-term research programme in Beatrix Bay by NIWA has focused on the issue of mussel farm sustainability, primarily considering the carrying capacity of growing areas for shellfish production (Gall *et al.* 2000; Gibbs *et al.* 2002; Safi & Gibbs 2003). This issue has been a major focus of attention during resource consenting and marine farm permit processes for mussel farms, and has mainly been driven by consideration of phytoplankton production and depletion (also referred to as food depletion) by cultured mussels. A key focus of this work has been on the 'ecological carrying capacity' of bays or regions for mussel culture. As part of our assessment, we also describe the role of water currents and waves in mitigating seabed and water column effects through flushing and dispersal of farm wastes. In this respect, an important effect of the farm structures is to alter natural flushing characteristics and hence the nature and magnitude of mussel farm effects.

In addition to the seabed and water column effects of mussel farms, a wider suite of ecological issues has emerged during resource consent and marine farm permitting processes, which are summarised in Figure 3. These include the effects of mussel farming on fish, seabirds and marine mammals, genetic interactions between cultured and wild Greenshell<sup>TM</sup> mussel stocks, and the potential for disease outbreaks. We address such issues in this report, and also recognise the role of mussel farms as artificial reefs, and related to this biosecurity risks such as the transfer of unwanted biofouling pests via movements of farm equipment and seed-stock. Biosecurity issues in relation to mussel aquaculture have gained recognition in the last decade, especially in relation to new large-scale farm developments around New Zealand, and as a result of incursions of relatively high profile fouling pests.





Figure 3. Schematic of potential ecological effects from long-line cultivation of Greenshell<sup>TM</sup> mussel and associated spat catching (modified from Forrest *et al.* 2007).



#### 2.3. Seabed effects of mussel cultivation

Seabed effects from mussel farms result from the sedimentation of organic-rich, fine-grained particles (mussel faeces and pseudofaeces), and the deposition and accumulation of live mussels, mussel shell litter and other biota attached to the ropes, floats and the mussels themselves. The predominant effect on the seabed typically arises from the deposition of faeces and pseudofaeces (referred to as 'biodeposits'), which leads to enrichment of the seabed due to the high organic content of the deposited particles. Mussel farms are almost invariably sited above soft-sediment habitats (as opposed to rocky habitats), hence information on seabed effects relates primarily to physico-chemical and ecological changes in those habitats.

#### 2.3.1. Biodeposition

The large body of international literature indicates that the main environmental impact of shellfish culture is increased sedimentation through biodeposition. Mussels filter particulate materials, primarily phytoplankton, but also zooplankton, organic detritus and inorganic sediment from the water. Particulate material is trapped in the labial palps of the shellfish, bound up with mucous, sorted and selectively ingested. Digestive wastes are later expelled as faecal pellets. Inedible or excess particulate material is loosely bound in mucous and expelled from the shell cavity as pseudofaeces. Faecal pellets and mucous-bound pseudofaeces (*i.e.* biodeposits) have greater sinking velocities than their constituent particles, thus mussel farms typically increase sedimentation rates under culture sites (Hatcher *et al.* 1994; Callier *et al.* 2006; Giles *et al.* 2006). In addition, detritus originating from epibiota attached to the culture structures contributes to the increased sedimentation (Kaiser *et al.* 1998). Sedimentation rates beneath mussel farms can vary with season (Giles *et al.* 2006), culture species (Jaramillo *et al.* 1992) and environmental conditions (*e.g.* tidal currents, water depth, riverine inputs), making monitoring of this process difficult.

#### 2.3.2. Changes to physico-chemical properties of the sediments

Numerous studies overseas and in New Zealand have documented changes to the physicochemical properties of sediments beneath mussel farms due to increased sedimentation and the accumulation of biodeposits (Dahlbäck & Gunnarsson 1981; Mattsson & Lindén 1983; Kaspar *et al.* 1985; De Jong 1994; Chamberlain *et al.* 2001; Giles *et al.* 2006; Callier *et al.* 2007; Hargrave *et al.* 2008). These include changes in sediment texture (Tenore *et al.* 1982; Kaspar *et al.* 1985; Stenton-Dozey *et al.* 2005); and local organic enrichment with an associated increase in oxygen consumption (Christensen *et al.* 2003; Giles *et al.* 2006), increased nitrogen release rates (Hatcher *et al.* 2004), sulphate reduction (Dahlbäck & Gunnarsson 1981) and lowered REDOX potential (Christensen *et al.* 2003; Grant *et al.* 2005).

For example, Giles *et al.* (2006) reported increased sedimentation rates under mussel farms in the Firth of Thames (New Zealand) relative to a reference site, with associated increased concentrations of organic carbon and increased sediment oxygen consumption within the

sediments. Similarly, Christensen *et al.* (2003) observed significantly higher ammonium fluxes and oxygen consumption (both evidence of high mineralisation rates) in sediments beneath a mussel farm in Beatrix Bay (Marlborough Sounds), resulting in increased sulphide levels in the sediments and a lower nitrogen removal rate (presumably due to impeded benthic photosynthesis from microalgae and reduced denitrification activity).

Monitoring of physico-chemical properties of sediments at mussel farming sites typically involves a suite of indicators; including observations of sediment colour (Figure 4), odour, REDOX potential discontinuity layer, sulphide concentrations and sediment organic content (Wildish *et al.* 1999). Of these indicators, sediment organic content has proven reliable and is often included (along with other indicators) in marine farm monitoring programmes in New Zealand and overseas. Elevated sediment organic content is commonly encountered beneath mussel farm sites in New Zealand. For example, Hartstein & Rowden (2004) found elevated levels (double that of reference location sediments) of sediment organic content at two sheltered mussel farm sites in the Marlborough Sounds. However, they observed levels beneath a mussel farm site located in a high energy environment to be similar to those observed in reference locations, highlighting how a dispersive environment can help reduce the level of seabed effects (Section 2.3.5).



**Figure 4.** Mud samples from beneath and adjacent an existing mussel farm site in Kenepuru Sound, Marlborough Sounds. Grey/brown coloured sediments indicate little or no organic enrichment, while dark grey/black sediments are indicative of reduced oxygen conditions.

Figure 5 summarises sediment organic content data collected by Cawthron during Fisheries Resource Impact Assessments (FRIAs) from directly beneath (Farm), adjacent to (Spreading) and outside the influence of (Reference) mussel farms in the Marlborough Sounds. Data from multiple assessments have been pooled to provide a general overview of the level of organic enrichment that can occur under standard Marlborough Sounds stocking densities (*e.g.* 150 mussels/m of crop rope). As Figure 5 illustrates, sediments directly beneath and within 50 m of the culture lines have, on average, slightly higher levels (~7.5% increase) of organic material (based on 1.5% increase in Ash Free Dry Weight, AFDW) than sediments outside the influence of the farms. In most cases, these slightly elevated levels of organic enrichment increase the productivity of coastal sediments without major disruption to community composition (see Section 2.3.3).



**Figure 5.** Organic content based on Ash Free Dry Weight (mean AFDW, + 1 SE) of sediments collected from beneath mussel farms (Farm), adjacent to farms (Spreading) and outside farm influences (Reference) in the Marlborough Sounds.

#### 2.3.3. Changes to biological properties of the sediments

Accumulation of organic matter and other associated changes in physico-chemical properties can create suboptimal conditions within the sediment matrix that can lead to changes in the abundance and diversity of micro- and macroscopic biota in the sediment (Danovaro *et al.* 2004 and references therein). For example, increased sedimentation beneath mussel farms can reduce microscopic plant production (Christensen *et al.* 2003; Giles *et al.* 2006), which can have a pronounced effect on oxygen conditions in the sediments and overlying water, as well as affect denitrification rates. Similarly, meiofaunal (very small organisms measuring a length of 0.45 to 1.0 mm) community composition can change significantly due to the presence of elevated organic content beneath mussel farm sites (Mirto *et al.* 2000).

However, the most widely used indicator of enrichment effects is sediment macrofauna (organisms measuring a length >1 mm) living within the sediment matrix, referred to as 'infauna'. According to models of organic enrichment (Pearson & Rosenberg 1978), sediments subject to increased organic loading will exhibit increased animal abundance, decreased species richness (number of different taxa) and animal biomass, and a shift in dominance of trophic groups (Weston 1990). Seabed enrichment selects for species more adaptable to low oxygen levels and/or to the instability of finer-textured, high organic sediments (Tenore *et al.* 1982).



Changes in physico-chemical characteristics beneath mussel farms can lead to a displacement of large-bodied macrofauna (*e.g.* heart urchins, brittle stars, large bivalves) and the proliferation of small-bodied disturbance-tolerant 'opportunistic' species such as capitellid polychaetes and other marine worms (Tenore *et al.* 1982; Mattsson & Lindén 1983; Kaspar *et al.* 1985; Christensen *et al.* 2003). The loss of large-bodied burrowing taxa can potentially have flow-on effects to sediment health due to a reduction in bioturbation and the associated irrigation of deeper sediments (Christensen *et al.* 2003).

In recent years, NIWA have used infaunal community composition to assess the level of seabed change at mussel farm sites in Wilson Bay, Firth of Thames, and have found "*little significant change*" in seabed community composition at sites monitored (Stenton-Dozey *et al.* 2004). These findings are consistent with numerous site assessments undertaken by NIWA in the Marlborough Sounds (NIWA unpublished data), where "*changes in the relative abundances of certain species rather than dramatic disappearances of intolerant species and appearances of new species*" have been observed (Stenton-Dozey *et al.* 2005).

We present here summary data describing changes to infaunal abundance and species richness at sites surveyed in the Marlborough Sounds during numerous Fisheries Resource Impact Assessments (FRIAs). As Figure 6 highlights, animal abundance was significantly higher at stations directly beneath marine farms when compared with spreading zone stations (sites within 50 m of a marine farm) and compared to reference sites. The way infaunal communities responded to farm-related enrichment was variable amongst sites, depending on environmental conditions such as depth and average current velocity (Hopkins *et al.* in prep). For example, species richness was either slightly depressed or slightly enhanced due to enrichment from farms, but averaged values were not appreciably different from reference sites (Figure 6).



**Figure 6.** Infaunal abundance and species richness (mean + 1 SE) within sediments collected from beneath mussel farms (Farm), adjacent to farms (Spreading) and outside farm influences (Reference) in the Marlborough Sounds (Hopkins *et al.* in prep).

In terms of infaunal community composition, Cawthron studies of sediment beneath mussel farms show an increase in the relative proportion of polychaete tubeworms (Figure 7), consistent with other published studies (Christensen *et al.* 2003; Stenton-Dozey *et al.* 2004). However, while the abundance of polychaetes was significantly higher beneath farm sites, the composition of other major infaunal groups (*e.g.* molluscs, crustaceans, echinoderms) was comparable between farmed and unfarmed locations, suggesting a low-level positive enrichment effect rather than a major disruption to the functional integrity of the sediments. However, it should be noted that these data represent an average across a wide range of water depths, water currents and farm ages. Higher level effects are likely to be observed when examining at farm sites that are predisposed to impacts (see Section 2.3.5).



**Figure 7.** Abundance of individual groups of taxa (mean + 1 SE) in sediments collected from beneath mussel farms (Farm), adjacent to farms (Spreading) and outside farm influences (Reference) in the Marlborough Sounds (Hopkins *et al.* in prep).

#### Effects on epibiota

New Zealand and overseas research to date has typically described ecological effects on the seabed based on infaunal communities as indicators. However, another important component of the seafloor community is the assemblage of animals and plants that live on the sediment surface, which are commonly referred to as 'epibiota'.

Depositional effects on plants and animals living on the surface of the seabed (referred to as 'epibiota') from mussel farms in New Zealand are not well documented. Although several studies have described reef-type communities (Kaspar *et al.* 1985; de Jong 1994) and an increase in predators (Kaspar *et al.* 1985; Grant *et al.* 1995; Inglis & Gust 2003) associated

with shell drop-off to the seabed below marine farming structures (described in following section). However, while several studies have described the attraction of epibenthic species to the food source provided by shell drop-off, few studies have quantified the displacement or destruction of epibiota beneath and immediately adjacent to mussel farms. One potential explanation for the paucity of information is the highly variable spatial and temporal abundances of epibiota. Such variability makes it difficult to attribute effects from mussel farms to epibiota distribution based on statistical comparisons. Several studies have observed the degradation of seagrass meadows beneath and adjacent to finfish farms (Pérez *et al.* 2008); however the relevance of these studies to mussel farms is questionable given the significantly higher organic loading experienced beneath finfish farms (Brown *et al.* 1987; Gowan & Bradbury 1987; Mirto *et al.* 2000; Crawford *et al.* 2003; Hopkins *et al.* 2004) and the fact that seagrass meadows in New Zealand are found primarily in intertidal and shallow subtidal areas.

In the absence of relevant published literature, we provide summary observations from a Cawthron investigation into tubeworm and red algae densities beneath mussel farm sites in Port Underwood, Marlborough Sounds. Polychaete tubeworms and red algae were observed on the seabed beneath and adjacent to proposed marine farm sites (Figure 8) during FRIA surveys in Port Underwood (Hopkins *et al.* 2005a,b,c). Polychaete tubeworms and macroalgae are not a common feature of the Marlborough Sounds region and are deemed to be of "special ecological value" (DoC 1995). In response to these findings, additional dive surveys were undertaken to determine the effect of mussel farming on the abundance and distribution of these ecologically important epibiota. Our investigations found much higher densities of tube worms (average cover ~25%) outside the boundaries of existing farms compared to beneath existing farms (average cover <5%). Densities of tubeworms were also observed to decrease in shallower water, which was probably related to changes in sediment composition closer to shore. Similarly, red macroalgae were generally more abundant outside marine farms (mean percent cover ~40%) than beneath existing marine farms (mean <5%).



**Figure 8.** Photos of red macroalgae and tubeworms observed on the seabed in Port Underwood, Marlborough Sounds.

NIWA monitored changes in mobile epibiota (*e.g.* snails, crabs) beneath mussel farms in the Firth of Thames during staged development in Wilson Bay (Stenton-Dozey *et al.* 2005), and

limits of acceptable change (LAC) were developed for various parameters (*e.g.* number of mobile epifauna, worm holes *etc.*) using baseline data collected prior to farm development. During the two years of farm development, LACs were exceeded along some transects for variables such as number of worm holes; however the LAC for the number of mobile epifauna was not exceeded (Stenton-Dozey *et al.* 2005). It was concluded that effects to such taxa from the mussel farm development were relatively small.

The above studies show that the significance of ecological effects from mussel farms is related to site-specific values, such as the presence of species or habitats that are sensitive to deposition or of special interest (*e.g.* high conservation value, keystone species). Ways to assess ecological values and determine locations for aquaculture development have been proposed elsewhere, for example in relation to mussel aquaculture expansion in the Marlborough Sounds (DOC 1995; Forrest 1995).

#### 2.3.4. Modification of benthic habitat

The most visually conspicuous effect to the seabed from mussel farming is the modification of the benthic habitat that can occur through accumulation of live and dead mussel material on the seafloor, produced primarily during harvesting and farm maintenance (Davidson 1998; Davidson & Brown 1999). Visual observations suggest that shell deposition within a farm can be patchy, ranging from rows of clumps of live mussels and shell litter directly beneath long-lines to widespread coverage across the farm site (Forrest & Barter 1999; authors' pers. obs.). Mussel clumps and shell litter beneath a mussel farm have been observed as acting as a substrate for the formation of reef-type communities (De Jong 1994; Davidson & Brown 1999). Kaspar *et al.* (1985) described reef-like communities under an existing farm that included large epibiota such as tunicates, sponges, sea cucumbers, calcareous polychaetes, and mobile predatory species such as starfish, crabs and fish. In other situations, mussel clumps and shell litter can remain relatively barren of reef-type communities (Watson 1996, Figure 9).

Several studies have described accumulations of scavengers attracted by mussel drop-off (de Jong 1994; Grant *et al.* 1995). It is likely that an increase in the numbers of predatory species will help to maintain a balance with respect to the large number of prey species (*i.e.* mussels). However, the potential concern is that the increased food source will create a predator oasis, which in turn may increase the potential for recruitment of juveniles into the adult predator population (Inglis & Gust 2003). Invertebrate predators such as the 11-armed sea star *Coscinasterias muricata*, aggregate beneath Greenshell mussel farms in New Zealand, where densities can be 39 times higher than at non-farmed sites (Inglis & Guts 2003; authors' pers. obs.). However, the link to increased recruitment has not been established. Theoretically, this potential increase of individuals into the adult population could also affect existing populations of benthic animals further away from the mussel farm. To our knowledge this has not been described existing mussel farming sites in New Zealand or overseas.





**Figure 9.** Shell debris beneath a mussel farm site in Kauauroa Bay (Marlborough Sounds), showing a sea cucumber foraging across shell litter (left), and macroalgae growing on live mussels deposited on the seabed (right).

#### 2.3.5. Factors affecting the spatial extent and magnitude of seabed effects

#### Magnitude of seabed effects

Available information for long-line mussel farms in both New Zealand and overseas (Dahlbäck & Gunnarsson 1981; Mattsson & Lindén 1983; Kaspar *et al.* 1985; De Jong 1994; Chamberlain *et al.* 2001; Grange 2002; Christensen *et al.* 2003) indicates that the spatial extent and magnitude of seabed effects depend on site-specific environmental characteristics (*e.g.* current speeds and directions, existing benthic habitat, wave climate, riverine influences, phytoplankton abundance), and to a lesser extent, farm management practices (*e.g.* stocking densities, line orientation, harvesting techniques).

The capacity of the environment to disperse and assimilate mussel farm biodeposition is largely determined by water depth and current speeds (*i.e.* flushing capacity), although the assimilative capacity of the environment may also vary seasonally in relation to factors such as water temperature. Increased flushing not only reduces localised sedimentation and accumulation of organic matter, but it also increases oxygen delivery to the sediments, allowing for more efficient breakdown (*i.e.* mineralisation) of organic material (Findlay & Watling 1997). For example, deep sites (>30 m) located in areas of strong water currents will have depositional footprints that are less intense and more widely dispersed than shallow, poorly flushed sites.

International studies show that the majority of environmental issues associated with biodeposition occur in systems where water exchange is restricted (Castel *et al.* 1989). Farm sites located in well-flushed tidal environments, as in the Spanish Rias, typically do not result in the accumulation of pseudofaeces but result in a favourable increase in macrofaunal biomass (Rodhouse & Roden 1987). However, where currents are very weak or water depth is shallow, biodeposition would be expected to contribute to hypoxic (reduced oxygen) conditions in the sediments. Such effects have been observed or inferred from models in sheltered embayments

or inlet systems (Dame & Prins 1997; Chamberlain *et al.* 2001; Grant *et al.* 2005; Waite *et al.* 2005; Cranford *et al.* 2007). For example, Dahlbäck & Gunnarsson (1981) describe excessive organic enrichment beneath mussel long-lines along the Swedish coast located in weak currents (3 cm/s) in relatively shallow water (8-13 m water depth), resulting in anoxic sediments with an overlying bacterial mat (*Beggiatoa* spp.) and significantly reduced infaunal biomass and diversity.

Culturing and husbandry techniques have also been identified as having the potential to influence the magnitude of seabed effects beneath a mussel farm; although very few studies have attempted to quantify this relationship. Miron *et al.* (2005) studied sediment beneath 19 blue mussel (*Mytilus edulis*) farms in eastern Canada. Their study found no strong relationship between environmental responses (*e.g.* organic matter and sulphide concentrations, REDOX profile or faunal diversity) and factors such as farm age and stocking densities; rather the environmental variables appeared to be correlated with the water depth at the site.

#### Spatial extent of deposition

Effects of biodeposits from mussel farms tend to be most evident directly beneath the long-line droppers; however a gradient of seabed effects has been measured at some farm sites (Hartstein & Rowden 2004), consistent with patterns of enrichment from other point source discharges (see Pearson & Rosenberg 1978). By contrast, live mussels, shell material and associated fouling biota have been observed to settle beneath the long-lines and are typically confined within 10 m of marine farming structures (Kaspar *et al.* 1985; Callier *et al.* 2007; authors' pers. obs.).

Cawthron has estimated the theoretical spatial extent of biodeposition for >50 proposed farm sites and extensions in the Marlborough Sounds using a simple depositional model which estimates the distance and direction pseudofaeces and faeces could travel before reaching the seabed. This model uses representative flow patterns and current speeds and an estimated particle-sinking velocity for faeces and pseudofaeces (Giles & Pilditch 2004; Hartstein & Rowden 2004). In areas of low flushing or shallow water depth, the spatial extent of biodeposition typically extended <50 m from the farm boundaries, while depositional footprints of >250 m were modelled for sites in more energetic environments or greater water depth (Hopkins *et al.* in prep).

These estimates are consistent with numerous assessments undertaken by NIWA in the Marlborough Sounds, where depositional effects footprints of 20-50 m were predicted for farms in small, sheltered embayments compared with footprints extended >200 m at sites with strong tidal forcing (Stenton-Dozey *et al.* 2008). Similarly, Hartstein & Stevens (2005) detected mussel biodeposits up to 30-50 m from mussel farm boundaries at sites located within a sheltered embayment.

The seabed environment beyond the effects footprint may be exposed to farm-derived materials, but has a capacity to assimilate them without exhibiting measurable ecological changes. It is conceivable, that in the future, more sensitive monitoring techniques (*e.g.* DNA


and genetic marking, stable isotopes, and digital sediment profile imagery techniques) may reliably detect these processes and effects further afield. But from an ecological perspective, the spatial extent of footprint associated with a typical mussel farms is considered well defined and predictable.

#### 2.3.6. Broader considerations of seabed effects

Deposition of fouling biota may also contribute to seabed enrichment beneath mussel farms. This situation may occur where fouling organisms reach high densities on farm structures and fall to the seabed either naturally or because of deliberate defouling by farm operators. The fouling biomass may intermittently be a substantial component of the organic material deposited to the seafloor, as appears to be the case for the recent spread of the invasive sea squirt *Didemnum vexillum* at mussel farms in the Marlborough Sounds (see Section 2.5.5). In such situations, the deposited fouling biomass may exacerbate enrichment effects (at least in the short-term) associated with other processes (*e.g.* biodeposition).

Finally, direct effects on the seabed from mussel farms could, under certain conditions, arise via processes other than deposition alone. For example, shading from farm structures could reduce the amount of light to the seafloor. This, in turn, could reduce the productivity of ecologically important primary producers such as benthic microalgae, or beds of macroalgae or seagrass, with a range of associated ecological effects (Huxham *et al.* 2006). Shading is unlikely to be a major consideration at present in New Zealand, but could conceivably arise if farms were located in environments where important primary producers are abundant directly beneath the farm structures. Shading effects are likely to be site-specific and can be effectively mitigated by appropriate farm site selection.

# 2.4. Water column effects of mussel cultivation

Effects of mussel cultivation on the water column are less well defined than for the seabed because they are inherently harder to quantify. By comparison, the water column is a highly dynamic environment that varies markedly in space and time due to complex hydrodynamics and the chemical and biological processes that occur within. This complexity is further compounded by the way that the mussel's physiological processes interact with the surrounding water. The following section summarises what is known about how mussels affect the physical, chemical and biological properties of the water column (in that order). Particular emphasis is given to the composition of the seston (which describes the state of the water column biology) and then to its wider implications for carrying capacity and sustainability.

# 2.4.1. Effects of farm structures on currents and waves

Currents generated by tides and/or waves play an important role in the transport and delivery of seston (drifting particles including phytoplankton) and dissolved nutrients gases (*i.e.*  $O_2$  and  $CO_2$ ), and the flushing of wastes and associated nutrients into and out of the marine system. Currents also influence seabed habitats and associated biota through sediment movement and shell litter deposition, and the flux of nutrients between the benthos and the overlying water column. If, for example, currents are not above a critical threshold to allow resuspension of seabed sediments and associated detrital material from shellfish farming, this could lead to excessive accumulation of organic wastes resulting in localised enrichment.

Currents are affected by marine farms due to drag forces that are created by the interaction of a moving fluid on an anchored submarine structure. The extent to which currents are modified will ultimately be dependent on the 'porosity' of the farm structures, or in other words, the extent to which the structures create drag and attenuate currents. The mechanism for this interaction has been well studied for a range of engineering applications (*e.g.* bridge supports in estuaries and rivers), but little research has been conducted in relation to marine farms. Our review of this field yielded literature only for scallop cages in China (Grant & Bacher 2001), mussel rafts in South Africa (Boyd & Heasman 1998) and long-line mussel farm structures in New Zealand (Plew 2005; Plew *et al.* 2005; Morrisey *et al.* 2006b; Stevens *et al.* 2008).

There have been two main approaches to assessing effects of farm structures: (i) measure and compare the differences in currents within and outside of existing farms (Boyd & Heasman 1998; Plew 2005), and (ii) estimate macro-scale changes using hydrodynamic modelling techniques (Grant & Bacher 2001; Stevens *et al.* 2008). Pioneering work by Boyd & Heasman (1998) on mussel rafts in South Africa showed decreases in current speeds within farms to be as little as 10% of the ambient flow. This study also investigated how changes in structural density (*i.e.* changes in porosity) affected currents within farms, revealing that increased rope density led to decreased current velocities. A more recent study by Plew *et al.* (2005) investigated changes in currents at a long-line mussel farm in New Zealand and found a 38% decrease in current speed and a reorientation of water flow parallel to the alignment of the mussel lines at peak velocities. Currents below the farm structure are often not affected by longline structures. In this regard, Plew (2005) notes higher currents "…beneath the farm than within, generating a shear layer below. However, the undercurrent was not significantly greater than velocities recorded upstream of the farm".

Two-dimensional (2D) hydrodynamic models applied by Grant & Bacher (2001) and Morrisey *et al.* (2006b) enable simulation of the effect of farm structures on currents, including deflection around (but not under or over) modelled structures. Given that in reality currents will also flow under structures (*e.g.* Plew 2005), the use of a vertically averaged 2D model represents a simplification of reality. Nevertheless, the results of these techniques applied to an intensively farmed open embayment in China, suggest a 54% reduction in current speeds inside the farmed areas and a 20% reduction within adjacent navigation channels (Grant & Bacher 2001). The authors also studied how these changes in currents would affect flushing, and found associated increases in flushing times (*i.e.* reduced flushing efficiencies) for this

intensively farmed bay. A New Zealand study on a less intensively farmed area estimated local changes in currents in the range of -10% around the majority of the farmed area to +20% in areas where current was deflected close to shore (Morrisey *et al.* 2006b).

Currents generated by waves can play a critical role in areas where tidal currents are too weak to provide adequate water movement. Episodic wave events in these situations may be critically important in facilitating resuspension and other mitigating processes (Panchang *et al.* 1997). The currents generated by a wave passing through a marine structure will interact with the structure in the same way as any other current; consequently a loss of wave energy will be seen as these interactions occur (Plew *et al.* 2005).

The depth at which currents are generated by waves depends on wavelength and period. Whether a given site will produce waves of sufficient magnitude and period to interact with the bottom will depend on site depth and wave climate, and the extent to which the wave climate is altered by farm structures. Longer period/wavelength waves contain more energy for the same amplitude wave and will produce currents to a deeper extent than short period waves. As a result, structures located close to the surface will attenuate energy from the short period waves more than long period waves. Evidence for this phenomenon is provided by Plew *et al.* (2005) where a mean energy attenuation of up to ~10% across a mussel farm structure was determined for short period waves (1-10 second). In relation to mussel farm site selection and mitigation of effects, minimising impacts on the wave climate would require locating structures in areas of long period wave action, *e.g.* wave-exposed coastal areas. Historically, this has not been the case as most farms were situated in embayments partially protected from wave exposure for engineering and logistical purposes.

Mussel aquaculture overseas is intensive by comparison with New Zealand and studies have shown that ecosystem function may be significantly affected by changes to currents and waves in coastal areas as development increases beyond critical levels (Grant & Bacher 2001). It should also be noted that some functions, such as seabed sediment movement, can be highly non-linear in their response around critical near-bed current values (Bridge & Dominic 1984; Dade 1993). For example, if near-bed current speeds were to halve, sediment suspension and transport may be reduced by more than half leading to an accumulation of sediments which may affect benthic organisms inhabiting these areas.

Despite evidence for local modification of currents and waves by farm structures, coastal ribbon development of marine farms in New Zealand is unlikely to significantly affect baywide hydrodynamic characteristics (Plew *et al.* 2005). While alteration of the wave climate shoreward of farms could theoretically affect ecologically important intertidal and shallow subtidal habitats (Davidson & Richards 2005), our observations at farm sites in the Marlborough Sounds provide no evidence to suggest that this is an issue at present levels of development (Section 2.3).

At the local farm-scale, effects on currents have potential implications for the sustainability of individual shellfish ventures and the local ecosystem. The structures themselves can increase

flushing times for an area, and in turn lead to an increase in localised seston depletion by allowing more time for the mussels to remove seston from the water. Hence, introduction of additional structures may also affect the growth rates of the culture and other organisms in an area that are also dependent on a steady supply of seston. The interaction between the influence of farms on currents and the ecological effects associated with seston removal is discussed further in Section 2.4.3.

# 2.4.2. Effects on seawater nutrient chemistry

Mussels and other associated fauna release dissolved sources of nitrogen (*e.g.* ammonium) directly into the water column as metabolic waste products. Water column nitrogen concentrations can also be increased due to enhanced benthic remineralisation rates beneath the farm (*i.e.* the microbial breakdown of mussel biodeposits on the sediment surface and flux of ammonium into the water column). This accelerated recycling of organic nitrogen in the seston provides a feedback mechanism that can stimulate further phytoplankton production thus counteracting seston depletion (Prins *et al.* 1998; Ogilvie *et al.* 2003). However, considering that the generation time (time for cells to double) for most phytoplankton is >1 day, any stimulatory response would likely occur outside the immediate growing area allowing sufficient mixing time to reduce nutrient concentrations to near ambient levels. Localised nutrient enrichment could more effectively stimulate production of algae attached to the mussels and culture lines (Black 2001). Tenore *et al.* (1982) speculated that such localised stimulation of algal production could potentially enhance coastal fish production.

Inorganic nitrogen is generally considered to be proportionally more limiting in temperate coastal waters than other nutrients that support phytoplankton production (Gibbs & Vant 1997; MacKenzie & Gillespie 1986). The amount of nitrogen removed from a mussel growing region via harvest is small in relation to the amount released to the environment as recycled nitrogen. Nevertheless, the amount exported through mussel harvest could be consequential if it is significant compared to the rate of replenishment from external marine and freshwater sources. In New Zealand growing areas, this situation is not likely to arise because mussel growth rates would not support an economically viable farm. For example, estimates for Pelorus Sound, an intensive growing region in the Marlborough Sounds, suggested that the annual nitrogen export via mussel harvest was <10% of the annual input from oceanic and freshwater inflows (calculated from Forrest *et al.* 2007).

Passage of water through a mussel farm could alter the dissolved oxygen composition of the water down-current from the farm. Oxygen is consumed through respiration by the mussels and associated fouling organisms on the culture lines. This can be exacerbated by enhanced benthic oxygen consumption due to deposition and decomposition of particulate organic materials beneath farms. To our knowledge, there have been no reports of the development of anoxic zones within the water column in New Zealand growing regions. This would be extremely unlikely unless farms were established in poorly flushed embayments, or at sites affected by enrichment effects due to other activities (*e.g.* fish farming).



#### 2.4.3. Seston removal and alterations

Long-line culture of filter-feeding Greenshell<sup>TM</sup> mussels effectively creates a fixed biological filtration system suspended through the upper 15 m or more of the water column. Although the clearance rate (the volume of seawater filtered by an individual mussel) can vary considerably according to mussel body size and seston quantity and quality, rates of up to 8.6 litres per hour have been reported by James *et al.* (2001). As such, a substantial proportion of the seawater flowing through a fully stocked farm can be "processed" by the mussels before moving beyond the farm boundaries.

During the mussel feeding process, particles are most efficiently extracted within an approximate size range of 5-200  $\mu$ m (Safi & Gibbs 2003), however particles as large as 600  $\mu$ m can be retained (Zeldis *et al.* 2004). This initial extraction can include phytoplankton, zooplankton (including copepods, fish and invertebrate eggs and larvae), protozoa, bacteria, detrital organic matter and inorganic sediment. Any fraction of ingested matter that is not assimilated may be discharged as faeces or pseudofaeces (Section 2.3.1). During the feeding process, mussels also consume oxygen and release dissolved nutrients into the water. Thus the composition of water passing through a mussel farm can be altered in a variety of ways; both in terms of the amount and composition of particulate matter as well as dissolved nutrients.

The extent to which a mussel farm removes seston from the water column is dependent on the ratio of the flushing time (which is affected by influence of structures on currents) to the rate at which the mussels filter and remove seston from the water (Gibbs 2007). The effect of introducing additional shellfish culture to an area will increase the removal rate through both the introduction of structures that increase the flushing time (due to current attenuation) and decrease the time available for the mussels to process the water as it passes through. Mussels will extract effectively less particulate matter from water that is more rapidly flushed through the farm than in situations where flushing is more restricted. In turn, the food available to the mussels is also less likely to become limiting when water is efficiently flushed through the farm. If significant food depletion occurs, cultured mussels could theoretically out-compete other suspension-feeders (*e.g.* zooplankton and benthic shellfish) for particulate food, or exceed what is termed the ecological carrying capacity of a farmed area (see Section 2.4.4).

Water column surveys conducted as part of FRIAs provide some evidence of phytoplankton depletion in and around existing mussel farms in New Zealand (misc. FRIA references). These surveys are temporal snapshots of phytoplankton abundance (as indicated by chlorophyll *a*), at a depth of ~3 m in and around the farms. The images can therefore be interpreted as a horizontal slice through the surveyed area 3 m beneath the water surface. In some instances, multiple surveys were conducted on a given day to capture different tidal states. Patterns of phytoplankton depletion were not always observed (see Figure 10 and Appendix 1) which reflects a high degree of temporal and spatial variability. Analysis of multiple surveys provides a general indication of the level of phytoplankton depletion that occurs from mussel farming.



Out of a total of 36 chlorophyll *a* (chl *a*) surveys conducted both within and outside of farmed areas, 21 had concentrations of chl *a* that were 1 to 15% lower within the farmed areas than outside of the farmed areas (see Table 9 in Appendix 1). The remaining 15 surveys revealed no differences or marginally higher chl *a* concentrations within the farmed areas. Based on these comparisons, it would appear that typical, small New Zealand mussel farms have relatively little influence on the overall concentration of phytoplankton in the water column, particularly within the context of the wider spatial area surrounding the farms (see Figure 10). It should be noted that while the overall abundance of phytoplankton may not change appreciably, these surveys do not account for effects of mussel feeding on the species composition of phytoplankton (*e.g.* two water samples may have similar chl *a* concentrations, but comprise substantially different phytoplankton assemblages - see following subsection).



**Figure 10.** Example FRIA surveys of chlorophyll *a* at 3 m depth in and around two mussel farming areas (delineated by lines) in Port Underwood. Two replicate surveys at the two locations are shown.

Modelled depletion shadows for proposed large-scale farm development sites have predicted reduced chl *a* concentrations extending beyond farm boundaries (Stenton-Dozey *et al.* 2008; Morrisey *et al.* 2006). As in the case of field surveys, model simulations indicate considerable variation in food depletion associated with environmental conditions (*e.g.* hydrodynamic patterns, background chl *a* concentrations *etc.*). Model outputs are typically conservative in that they assume access to the entire volume of water in a model cell (*i.e.* perfect mixing), whereas in reality these animals have restricted access to the water surrounding them. Despite these assumptions the results of these models for present level of development are generally not sufficient to result in adverse ecosystem effects over bay-wide scales. In support of this conclusion, configuration of a linear carrying capacity model (Ecopath) for various mussel aquaculture scenarios in Tasman and Golden Bays (Jiang & Gibbs 2005) suggested that the development of proposed large aquaculture management areas would not significantly alter the ecological structure of the food web. Carrying-capacity issues are discussed further in Section 2.4.4.

Predictions of the extent and intensity of food depletion effects for various proposed largescale mussel farm developments generally agree that mussel farming can lead to measurable water column effects at a local farm scale, but that significant alteration of ecosystem characteristics would be unlikely. An assumption enabling this generalised conclusion is that farms are typically located where adequate flushing occurs. Zeldis *et al.* (2008) conclude that climatic forcing conditions (*i.e.* SOI and associated oceanographic states and weather patterns) largely control inter-annual variability in phytoplankton biomass and mussel yield in Pelorus Sound; an intensively farmed region of the Marlborough Sounds. Their results indicate that, contrary to some suggestions, reduced mussel yields in 1999-2002 were not related to particulate food depletion by cultured mussels.

Seston removal by cultured bivalves has been considered by some to be an example of topdown control that could have beneficial environmental effects through amelioration of eutrophication effects and improvement in water clarity (Officer *et al.* 1982; Gottlieb & Schweighofer 1996). Others dispute this because most of the ingested organic material would be rapidly recycled into the water column as inorganic nutrients to stimulate phytoplankton production. Therefore the net effect on phytoplankton dynamics could be to increase turnover and overall production rather than limit phytoplankton biomass (Nizzoli *et al.* 2005). See below for further discussion of this topic.

Finally, while most of this section considers plankton alteration due to filtration and seston removal, it is worth acknowledging that mussels can also contribute significantly to the seston. During spawning events mussels release a significant portion of their biomass as gametes into the water column where the larvae develop and metamorphose before attaching to substrates as mussel spat. As an example, it has been estimated that an average farm can lose as much as 50% of its crop biomass (wet weight) within a few hours over a single spawning event (K Heasman, pers. comm.) and an average farm can hold 100-150 T mussels/ha. The ecological ramifications of this large pulse of organic material into the water column remain undescribed.



#### Alteration of plankton community structure

There is a possibility that passage of water through a mussel farm could alter the plankton community structure down-current from the farm, however the degree to which this occurs in New Zealand growing waters (or the ecosystem implications thereof) is yet to be properly evaluated. A number of studies suggest that food items may be specifically selected by some bivalve species, based on particle size and/or nutritional value (Bourgrier *et al.* 1997; Shumway *et al.* 1985). Selection of phytoplankton according to size class has also been reported for *P. canaliculus* by Safi & Gibbs (2003) who noted that mussels are unable to efficiently capture phytoplankton cells <2  $\mu$ m in size. Thus the small-celled picoplankton, which can comprise a significant proportion of the phytoplankton community, may not be removed by the mussels, and water passing through a farm might be expected to contain a higher proportion of picoplankton compared to the larger size classes that are preferentially removed. Therefore it seems likely that preferential filtering may result in changes to the size structure of the plankton communities in a farmed area, particularly in areas of low flow. However, as stated above further research is required to confirm this.

It is appropriate to note here, also, that shellfish populations have the potential to influence the animal component of the plankton (or 'zooplankton'). This relationship was first highlighted Zeldis *et al.* (2004) in a study that established clearance rates of adult, copepodite, and naupliar copepod stages by mussels (20, 31, and 49 l individual<sup>-1</sup> day<sup>-1</sup>, respectively). Most zooplankton are generally considered too large to be utilised by mussels, however there has been some concerns raised over the ability of shellfish to consume planktonic larvae, and in particular, the larvae of fish. Knowledge relating to this issue is discussed in more detail in Section 2.5.2.

# Harmful algal blooms

Harmful algal blooms (HABs) represent a particular risk in mussel growing waters. Although such blooms may be influenced by seawater nutrient concentrations, there is no evidence to indicate that localised farm-generated enrichment or alteration of phytoplankton communities (as discussed above) have resulted in an increased incidence of HABs. It is also important to recognise that toxic algae blooms can be a natural phenomenon and occur near-annually in regions of New Zealand that do not have established shellfish farms, *e.g.* BOPP and Hawke Bay (Keeley *et al.* 2005).

# Table 3.Incidences of 'high' and 'very-high' HAB risk from total phytoplankton biomass, and species that<br/>are ichthyotoxic or toxic in shellfish from 30 sites in the Marlborough Sounds (Figure 11).<br/>Summary of data supplied by Marlborough Shellfish Quality Programme (MSQP).

	Toxic in shellfish (%)		Ichthyotoxic species (%)		Phytoplankton biomass (%)	
<b>Risk category:</b>	High	V. high	High	V. high	High	V. high
Total incidence from all 30 sites	3.04	0.97	0.14	0.03	0.64	0.32



Phytoplankton monitoring is presently carried out in all aquaculture regions in New Zealand, including ~30 sites situated around the Marlborough Sounds. A recent appraisal of eight years worth of the Marlborough Sounds data collected by Marlborough Shellfish Quality Program (MSQP), revealed very low incidences of elevated HAB risks. Risk specifically from phytoplankton species toxic to humans consuming shellfish were assessed to be 'High' and 'very high' on 3% and <1% of occasions, respectively (Table 3, Figure 11). Incidences of elevated risk from ichthyotoxic species were even less common, with <1% of all samples scoring a '>/=High' ranking. There are also no obvious spatial patterns consistent with the distribution of mussel farms within the region (Figure 11). Included amongst the high risk species were ASP-producing *Pseudo-nitzschia* spp., as well as NSP-producing *Karenia* cf *mikimotoi* and *Heterosigma akashiwo* which are both toxic to fish. Hence, incidences of HABs within New Zealand's main mussel growing area have been generally low.





**Figure 11.** Graduated circle plots of % incidence of 'high' (left) and 'very high' (right) HAB risk at MSQP monitoring sites in the Marlborough Sounds. A & B. Phytoplankton biomass, C & D. ichthyotoxic species, and E & F. phytoplankton toxic in shellfish.



# 2.4.4. Ecological carrying capacity

As described above, mussel farms act as biological filters and influence the types and amount of food available in the water column. This in turn has the potential to have top-down effects on the wider ecosystem by influencing the amount of resources available at the base of the food web. Due to high density cultures and relatively high filtration capacity of mussels, the concept of carrying capacity has been focused on the depletion of food within the water column. Questions around the concept of carrying capacity arose following observed decreases of ~25% in mussel yields in the Pelorus Sound during 1999-2002 (Zeldis *et al.* 2008). The observed reduction in yields coincided with increased demand for water space for shellfish culture, which in turn lead to research effort being directed toward the question of what level of culture is sustainable (*i.e.* carrying capacity) in the growing areas. In order to understand the magnitude of the effects of the farms, bio-physical models have been used to assist in understanding the cumulative spatial impacts of mussel farms on these primary resources in Pelorus Sound (Ross *et al.* 1999; Inglis *et al.* 2000, Zeldis *et al.* 2008), Bay of Plenty (Longdill *et al.* 2006) and the Coromandel (Broekhuizen *et al.* 2002; Stenton-Dozey *et al.* 2008).

Typically these studies have focused on the levels of culture that reduce the food in the water to concentrations where they begin to affect the growth of the culture itself. This approach relates to production carrying capacity (*i.e. the stocking density of bivalves at which harvests are maximised* – Inglis *et al* .2000) or the physical carrying capacity of a given coastal area (*i.e. the total area of marine farms that can be accommodated in the available physical space* – Inglis *et al* .2000). Carrying capacity of mussel farming can also be defined in relation to ecological effects, or what is termed ecological carrying capacity.

Ecological carrying capacity has been defined for shellfish aquaculture by Inglis *et al.* (2000) as "*the stocking or farm density which causes unacceptable ecological impacts*" and more recently by Gibbs (2007) as the "*level of culture that can be supported without leading to significant changes to ecological processes, species, population or communities in the growing environment*". However, it seems that even small scale developments will have some effect on "*ecological processes, species, population or communities in the growing environment*". Whether or not these effects are *unacceptable* or *significant* and exactly what defines the *growing environment* (*i.e.* at what scale do we look for effects; at the physical footprint of the farm or bay-wide?) is still a source of debate in the scientific, policy and social arenas.

A lack of formal guidelines to assist stakeholders and scientists as to what are acceptable changes to environmental parameters may be part of the reason that studies have typically focussed on production carrying capacity to determine acceptable levels of culture. However, the development of guidelines is not a simple task given that each growing area will be unique and development may cause a number of potentially unacceptable impacts, namely:

- Effects on suspended algal concentrations and production (Gibbs 2007)
- Effects to the recruitment of juvenile larvae (Gibbs 2004)
- Changes to the benthic and pelagic habitat (Gibbs 2004)



- Changes to predator-prey relationships (Gibbs 2004)
- Changes to physical properties of the system (*e.g.* currents, flushing rates, waves and density structure Plew 2005)

The likelihood of any cumulative effects of shellfish aquaculture on these properties will be dependent on the size of the culture, the sensitivity of the organisms in the ecosystem and the proximity of the system to any perceived "tipping points" and would need to be assessed on a case by case basis. Gibbs (2007) provides some guidance as to what sustainability performance indicators may be acceptable for assessing the level of interaction between bivalve culture and the water column environment. These can be applied to various farm locations as a means of identifying any limitations relating to existing environmental characteristics (*e.g.* hydrodynamics, phytoplankton biomass *etc*). For example, he suggests that the time for the water in an area to be replenished (flushing time) should not exceed 5% of the clearance time (filtering efficiency) of the mussels in order to reduce the likelihood of exceeding ecological carrying capacity.

There is also on-going development and application of models that consider the broader ecosystem effects of mussel farms. Application of food web models assist in estimating and forecasting the range of possible cumulative effects to higher trophic levels that would otherwise be too difficult to quantify based on field measurements. For example, the ECOPATH steady-state mass balance model (Christensen *et. al.* 2000) was recently applied to assess the potential of Tasman Bay for mussel aquaculture development (Jiang & Gibbs 2005). The model was used to determine an ecologically sustainable level of mussel biomass beyond which higher trophic levels of the ecosystem might be affected through competition for finite energy resources (*i.e.* food). Although this study focused on higher trophic levels in a general sense, the technique also has the ability to focus on a particular high-value species if it was deemed important.

The results of Jiang & Gibbs (2005) indicate that significant ecosystem energy flow changes occurred at mussel biomass levels less than 20% of a mussel dominated ecosystem, thus implying that ecological carrying capacity limits may be much lower than production carrying capacity limits. However, the a spatial approach taken by these techniques limits the findings to a broad, bay-wide scale assessment of ecological carrying capacity and does not incorporate feedback mechanisms such as changes to the flushing regimes induced by structures (Grant & Bacher 2001; Plew *et al.* 2005) or far-field nutrient enhancement and increased phytoplankton growth (Gibbs *et al.* 1992).

#### The New Zealand situation

So where does New Zealand sit currently in terms of meeting the objectives of ecological carrying capacity? Unfortunately, there are no definitive studies which provide a clear cut answer to this question, mostly because it is complex issue. In order to consider it, we first need to determine the temporal and spatial scales to be assessed. Typically the results of studies conducted as part of consent applications for individual farms (*e.g.* Pelorus Sound – unpublished Cawthron FRIA Reports, and the Coromandel - Stenton-Dozey *et al.* 2008)

suggest that the current levels of production are presently low when compared to average levels of food in predominantly semi-confined growing regions (*i.e.* Embayments/Sounds). Despite the reduced production noted over 1999-2002 (Zeldis *et al.* 2008), the conclusions of these studies are supported by a generally consistent production of mussel culture over the longer term, suggesting New Zealand mussel farms are at least sustainable in a production sense.

Very high development is present in some countries where the level of culture is only limited by the physical carrying capacity (*e.g.* Sungo Bay, China – Bacher *et al.* 1998). By comparison, development of shellfish farming in New Zealand coastal waters is relatively conservative in the sense that growing areas are unlikely to reach production limits due to other constraints on development (*e.g.* social constraints; Banta & Gibbs 2007). Despite this relatively low level of culture, bad production years have been noted, which Zeldis *et al.* (2008) attribute to climatic forcing conditions and inter-annual variability in phytoplankton biomass over multi-year time scales. This suggests that some areas may operate close to sustainable production limits during years of naturally low primary production. Ecological carrying capacity limits are likely to be lower than production limits (Jiang & Gibbs 2005), so it would follow that ecological carrying capacity may have been periodically exceeded by the current level of culture in New Zealand. It is also possible however, that mussel growth (and production) may have slowed during low productivity years irrespective of farming intensity, and this is one of the difficulties with attributing lower production to an unsustainable farming intensity.

Exceeding ecological carrying capacity may be acceptable if it is confined to an acceptable scale of impact (*e.g.* within farm limits or a small fraction of an ecosystem). Typically a baywide spatial scale is used to define an acceptable level of change (Jiang & Gibbs 2005; Inglis *et al.* 2000; misc. FRIA reports). This seems appropriate given that this physical delimitation of the ecosystem makes it possible to account for growth, transport and decay of mobile resources (*i.e.* phytoplankton) and changes to significant physical characteristics (*e.g.* flushing rates). Growing areas currently represent a small fraction of the total area of embayments in New Zealand (*e.g.* the heavily farmed Beatrix Bay has approximately 5% of the bay area under culture); hence it seems unlikely that bay-wide scale breaches of ecological carrying capacity are likely to have occurred as a result of the present level of culture.

Spatial modelling tools offer a way of estimating the extent to which the cumulative effects of mussel farming may be approaching ecological carrying on "bay-wide" and "regional" scales. However, knowledge gaps are still evident in these models; particularly in the biological aspects (*e.g.* feeding behaviour and growth of the shellfish - Heasman *et al.* in prep) which are still areas of active research. Long-term *in situ* monitoring of important ecosystem parameters may be necessary if models determine that a development is close to exceeding ecological carrying capacity.



# 2.5. Wider ecological effects from Greenshell<sup>™</sup> mussel cultivation

#### 2.5.1. Habitat creation

Marine farms and other artificial structures in marine environments provide a threedimensional reef habitat for colonisation by fouling organisms and associated biota (Costa-Pierce & Bridger 2002). By comparison with natural rocky or soft-sediment habitats, such structures can provide a substantial surface area for the attachment of biofouling. Studies from New Zealand (*e.g.* MAF Biosecurity New Zealand port baseline surveys) and overseas (Hughes *et al.* 2005; Braithwaite *et al.* 2007) indicate that the dominant biota on such structures includes macroalgae (seaweeds) and sessile (attached) filter-feeding invertebrates such as sea squirts, bryozoans and mussels. These assemblages typically have a range of other non-sessile animals associated with them, such as polychaete worms and various small crustaceans. Based on overseas research, it is evident that the assemblages that develop on artificial structures can be quite different to those in adjacent rocky areas (Glasby 1999; Connell 2000).

Artificial structures are recognised as providing novel foraging habitat, detrital food sources, breeding habitat, and refuge from predators for some species (Dealteris et al. 2004). The significant filtration and biodeposition capacity of sessile filter-feeding communities associated with artificial structures is also well recognised (Hughes et al. 2005), but the ecosystem effects of such processes are not well understood. In relation to mussel farms in New Zealand, the functional role of the associated fouling community is not well understood, but we would expect it to contribute in some way to the water column and seabed effects that were described above. Overseas studies show that the filtration capacity of extensive fouling communities has the potential to deplete phytoplankton and other particulates from the water column (Mazouni et al. 2001). In addition, biodeposits (i.e. faeces from consumed food and pseudofaeces from unprocessed food) produced by the fouling community, and inadvertent or deliberate removal of fouling biomass, have the potential to exacerbate seabed enrichment. Introducing significant quantities of artificial reef structures in the form of marine farms with the associated fouling assemblages are also thought to influence fish assemblages (see Section 2.5.2). Some of the other wider implications of the novel habitat created by mussel farms are outlined in subsequent sections.

# 2.5.2. Effects on fish

Unlike the literature for finfish farms, studies which describe how mussel farms affect wild fish assemblages are sparse. This comparative dearth of information is probably consistent with a general lack of concern over the potential for adverse effects. The main difference between these two forms of aquaculture is that fish farming involves an external feed input, which can be a strong fish attractant and increase wild fish abundances and productivity (Dealteris *et al.* 2004; Dempster *et al.* 2004; Machais *et al.* 2004, 2005, Dempster *et al.* 2006). However, some of the mechanisms by which fish are affected still apply and are discussed here. Firstly, mussel farming still involves introducing a complex three-dimensional structure

to an otherwise featureless seabed (*i.e.* sand/mud), which can be colonised by a diverse and productive fouling community. Such alterations to the existing habitat in turn alter the environments suitability to fish (Caselle *et al.* 2002; Dempster *et al.* 2006). This is the same premise upon which FAD's (fish attraction devises) are used to aggregate fish for commercial and recreational fishing purposes (Buckley *et al.* 1989; Relini *et al.* 2000; Dempster & Kingsford 2003). Hence, it is commonly believed that marine farms have the propensity to enhance fish abundances (Dealteris *et al.* 2004).

Wild fish abundances can also be affected by changes in the way the area is subjected to fishing pressure. Sites that have historically been part of a trawl fishery for example, will no longer be accessible due to the presence of structures, essentially creating a commercial 'notake' area akin to a marine reserve (Dempster et al. 2006). This issue is particularly true in the case of very large offshore sites that may occupy space in historically fished areas (e.g. Hawke Bay AMA). Influences from removing commercial fishing pressure may however be somewhat offset by changes in the way the area is utilised by recreational fishers (authors', pers. obs.). In the Coromandel for example, marine farms are generally viewed as good fishing locations, particularly when crop is being harvested and the fouling organisms that are being cleaned from the mussels are being discharged back into the water. It is presently unknown whether this increased recreational pressure around marine farms has a negative effect on the wider fish population. If the farms are aggregating fish from a wider area, but not enhancing the populations, then the stocks may be reduced; or alternatively, if the farms are providing additional food and habitat then the population may be enhanced, and the increased recreational pressure sustainable. It is worth noting also that the creation of a FAD-like structure also has the potential to alter existing fish assemblages by favouring particular species.

Some studies also warn against presuming that artificial structures constitute effective fish habitat. One such study, which compared natural to artificial reefs (Clynick *et al.* 2008) found that while they supported similar species, the overall assemblages were quite different. Likewise, a study beneath mussel farms in the northern South Island found that the farm structure did not necessarily provide habitat for significant numbers of commercially or recreationally important fish (Morrisey *et al.* 2006). Instead, the farms were associated with species characteristic of local demersal habitat. This raises an important point; any effects are likely to be site- and region-specific due to the different fish species that may be present, and each species may have unique responses to the type of artificial habitat (Morrisey *et al.* 2006). The precise effect on wild fish assemblages will therefore be difficult to predict without reference to a comparable scale operation within the proposed bay/region.

The most comprehensive New Zealand study to date was conducted by Morrisey *et al.* (2006), which compared fish assemblages around farm droppers with those inshore in Golden Bay and the Marlborough Sounds. As well as noting low abundances of commercial species, that study described generally low abundances and communities dominated by small demersal fish species, especially triplefins (*Forsterygion lapillum* and *Grahamina gymota*) and wrasse (or 'spotty') *Notolabrus celidotus (e.g.* Figure 12-D). Similarly, an earlier study in the same

region established a positive correlation between young spotties and alagal cover on mussel lines, and with encrusting biota on mooring ropes and blocks (Carbines 1993). The only other significant study in the field (also from the Marlborough Sounds), added that mussel farms were associated with more and a greater variety of fish species (Grange 2002). Most notably, the elevated fish taxa included a valuable commercial species, blue cod; however, none of the differences were statistically significant (Grange 2002). Larger pelagic species, such as kingfish, kowhai and yellow-eyed mullet showed no obvious preference towards farm structures. Although this may be the case in inshore areas, observations from around offshore farms in Hawke Bay and Opotiki suggest that pelagic species are more common around structures than in open water (when they are present during summer, Figure 12-A, -E, Keeley N, pers. obs.). Other observations from around offshore aquaculture structures have identified small groups of juvenile leather jackets (Parika scaber, Figure 12-F) and trevally (Caranx geogians, Figure 12-C) seeking shelter within the lines, and of adult red cod (Pseudophycis bachus) occupying spaces between cages nearer the seabed (Figure 12-B). This is a clear example of the potential for site- and region-specific differences in the effects on fish assemblages.

# Larval grazing by shellfish

Gibbs (2004) noted that cultured shellfish populations also had the potential to directly reduce recruitment into fishery populations through the consumption of eggs and larvae as observed by Davenport *et al.* (2000) and Lehane & Davenport (2002). Although no field research has been undertaken to assess the extent of this grazing mortality, blue cod recruitment (an abundant species in the area) in Admiralty Bay, where mussel culture occupied about 10% of the total bay area, was assessed in a desktop study (Gibbs 2004). The results suggested that the impact from the level of culture (as at 2004) is equivalent to additional mortality (on top of variable natural mortality) of less than 10%. The study also notes that this reduction could be negated by allowing a further 1.1% of the female spawning stock to remain unfished. In a similar study, Broekhuizen *et al.* (2004) modelled the possible effects of a large farm development in the Firth of Thames on survival of snapper (*Chrysophrys auratus*) larvae. They concluded that the farm could reduce numbers of eggs surviving to age 8 days by 2.5-15 % when fully developed (>2000 ha), and by 2-6 % with the existing level of farms.

Consequently, the sparse litterature suggests that the grazing influence of farms could have an impact on recruitment to fisheries; the scale of this effect will largely be governed by the extent of the culture, behaviour of larvae and flow dynamics of the region in question. In the case of the Firth of Thames, this finding was significant because the farm development concerned is thought to be situated near to important snapper spawning grounds. However, given Admiralty Bay represents one of the most intensively farmed regions in New Zealand, impacts greater than the 10% determined by Gibbs (2009) seem unlikely under present culture pressures. It should also be noted that both of the desk-top studies for depletion estimates cited here, potentially overestimate grazing for a number of reasons. Firstly, they assume perfect mixing within modelled "cells" of water and that there is no avoidance ability on behalf of the larvae, nor any size selection preference being exhibited by mussels. The proportion of eggs





encountered is also out of the total released and needs to be put into context with anticipated natural mortality.

Figure 12. A: kingfish schooling on surface above mussel farm research structures in Opotiki, B: Red cod seeking shelter between cages at 30 m depth on a mussel research site in Hawke Bay, C: juvenile trevally amongst a mussel research site in Hawke Bay, D: wrasse (spotties) amongst mussel lines in Marlborough Sounds, E: kingfish schooling around mussel research lines in Hawke Bay, F: leatherjacket around mussel rope.



# 2.5.3. Effects on seabirds

#### Overview

Several New Zealand and overseas studies discuss the potential ecological effects of shellfish aquaculture on seabird populations, but only a few direct studies have been conducted (Roycroft *et al.* 2004; Zydelis *et al.* 2006; Kirk *et al.* 2007). Based on these studies, mussel aquaculture potentially affects seabirds by altering their food resources, causing physical disturbances (*e.g.* noise) and/or being a possible entanglement risk. The structures associated with aquaculture may also provide benefits including additional perching and feeding opportunities. As several of New Zealand's seabird species are endangered or threatened, it is important that the shellfish industry remains up-to-date on any possible influences shellfish farming may have on these populations (Dowding & Murphy 2001).

# Effects on food supply

The attraction of certain seabird species to aquaculture structures has been noted within New Zealand and overseas aquaculture regions (Ross *et al.* 2001; Butler 2003; Roycroft *et al.* 2004). An increased abundance of seabirds associated with shellfish farms may indicate increased fish abundance within the sites, as aquaculture structures are thought to provide alternative or additional habitats for some fish species (Costa-Pierce & Bridger 2002; see Section 2.5.2). In New Zealand, shags (little, pied, spotted and king), terns (white fronted) and gulls (black-back and red-billed) have been observed foraging in and around mussel farms (Brown 2001; Lalas 2001; Butler 2003). An Irish study found significantly more piscivorous seabird groups (*e.g.* shags, guillemots, razorbills and gulls) utilising mussel farm sites compared to control areas (Roycroft *et al.* 2004), suggesting birds were benefiting from elevated fish numbers in farm areas.

From overseas studies, several seabird species (*e.g.* oystercatchers, plovers, gulls) are known to feed directly on shellfish stocks or associated fouling biota (Ross *et al.* 2001; Roycroft *et al.* 2004; Kirk *et al.* 2007). For example, Kirk *et al.* (2007) noted that farmed mussels tended to be larger, thinner-shelled and attached more weakly than wild mussels, and along with higher stocking densities these characteristics may attract avian predators to a farm. Currently, there are no reports of New Zealand seabirds foraging directly on shellfish farm stock.

Overall, New Zealand (Butler 2003) and overseas (Ross *et al.* 2001; Roycroft *et al.* 2004; Kirk *et al.* 2007) studies suggest that the general attraction of particular seabirds to mussel farms is likely due to increased foraging success on fish and biofouling, and even on the cultured stock itself. The consequences of this attraction will likely depend on the species' dietary preferences and response to both direct and indirect ecosystem changes induced by mussel cultivation.

# Human disturbance

Several studies have noted that certain seabird species are relatively sensitive to human presence and disturbance (Goss-Custard & Verboven 1993; Butler 2003; Roycroft *et al.* 2004, Gross-Custard *et al.* 2004). Butler (2003) found that nesting king shags in the Marlborough Sounds were highly susceptible to disturbance by boats and even aircraft, leading to part or

complete abandonment of nests and chicks. Such findings suggest that day-to-day maintenance, harvesting and other activities taking place on mussel farms located near breeding or roosting sites have the potential to adversely affect bird populations, possibly in tandem with other sources of disturbance (*e.g.* recreational vessel activities). Nonetheless there is no direct evidence for such effects from mussel farming in New Zealand, and the little available information is inconclusive. For example, varying levels of responses to boat disturbance have been reported for several different shag species in New Zealand depending on location and their activity state (Brown 2001; Lalas 2001). However, Lalas (2001) concluded that boat traffic alone was not a significant disturbance factor to king shag foraging and/or resting activities.

During certain times of the year (especially summer) seabirds may use farm structures as perching sites as look-outs (*i.e.* to spot fish) or to evade shore predators and avoid human disturbance on shore. Roycroft *et al.* (2004) found evidence of shorebirds as well as more terrestrial species, such as oystercatchers and plovers, using farm structures rather than more traditional land sites in Ireland. Despite their apparent sensitivity to disturbance, New Zealand king shags have been observed regularly roosting on mussel buoys (Brown 2001; Lalas 2001). Brown's (2001) results suggest that king shags favour roosting on floats over land sites. Overall, the potential disturbance of seabirds from nearby mussel farms appears to be dependent on the bird species, farm location in relation to nesting or breeding sites, and the relative disturbance of farm operations (*e.g.* noise and boat traffic) in comparison to other local forms of disturbance (*e.g.* recreational boating, casual or commercial use of nearby beaches).

# Entanglement

Incidental seabird entanglement in fishing gear, including set-netting, line fishing and even crayfish pots (Butler 2003; Bull 2007), are a recognised national and international problem to seabird survival (Taylor 2000a, 2000b). However, to date, no entanglements of seabirds in New Zealand or overseas shellfish farm lines have been reported. As with marine mammals (see Section 2.5.4), loose and more fibrous thin lines pose the greatest threat to diving seabirds. Hence, entanglement risk appears low in the New Zealand mussel industry where the long-line method places lines under considerable tension.

A potentially greater risk within the mussel industry is operational by-products of farms, including lost lines and plastics (Weeber & Gibbs 1998). The mussel industry has an Environmental Code of Practice that seeks to minimise such risks, and they are likely to be minimal to non-existent in well-maintained farms. The potential is considered to be greater, however, after stormy weather (Page *et al.* 2000) and in poorly operated farms. Butler (2003) found young and adult Australian gannets (*Sula serrator*) in the Marlborough Sounds entangled in discarded rope ties from mussel farms that had been incorporated into nests by parents. In an overseas example, ingestion of plastic debris by albatross chicks caused mortality through dehydration, gut blockage and/or toxic effects during digestion (Auman *et al.* 1998).



# 2.5.4. Effects on marine mammals: seals, dolphins and whales

#### Overview

Interactions between marine mammals and aquaculture usually result from an overlap between the spatial location of the facilities and the breeding, feeding and/or migrating habitat of the species. Several New Zealand (Slooten *et al.* 2001; Markowitz *et al.* 2004) and overseas (Würsig & Gailey 2002; Kemper *et al.* 2003; Watson-Capps & Mann 2005; Wright 2008) studies have characterised the interactions arising in these cases, which include competition for space (habitat exclusion), underwater noise disturbance, potential for entanglement and flow-on effects due to alterations in trophic pathways. Our discussion below shows that the implications of direct interactions between marine mammals and shellfish aquaculture have been fairly well examined. To date, issues such as habitat exclusion, underwater noise and entanglement appear to be minor for New Zealand mussel farming, although the potential for adverse effects still exists with continued growth in both marine mammal populations and the industry. Potential risks are best identified and managed on a case-by-case basis; for example by selecting farm locations to minimise the likelihood of overlap with marine mammal migration routes and/or known habitats.

#### Habitat exclusion

Mussel farm droppers typically extend vertically from floats at the surface through the water column to within a short distance (c. 1-5 m) above the seabed (see Figure 3); although some farms within the Marlborough Sounds only occupy the top half of the water column at a 30-40 m deep site. As Markowitz et al. (2004) demonstrated with sonar, such vertical structures may appear as visual or acoustic three-dimensional barriers that can potentially exclude marine mammals from habitats previously used for feeding, calving and/or migration activities. Studies in New Zealand have so far only addressed interactions between mussel farms and Hectors and Dusky dolphins. A study by Slooten et al. (2001) confirmed only one sighting of a Hector's dolphin (Cephalorhynchus hectori hectori) within boat access lanes, not between the actual lines, of a farm in Golden Bay. While this observation alone does not provide compelling evidence for an exclusion effect, multiple studies in Admiralty Bay (Marlborough Sounds) have observed significantly fewer dusky dolphins (Lagenorhynchus obscurus) inside mussel farms than outside (Markowitz et al. 2004, Vaughn & Wursig 2006, Duprey 2007, Pearson et al. 2007). Collectively this work suggests that while some marine mammal species are not completely displaced from regions as a whole, they do not appear to be utilising habitats occupied by shellfish farms in the same manner as prior to the farms' establishment.

Overseas research highlights that the nature of habitat exclusion greatly depends on the type of culture method and the particular species of marine mammal present in the cultivation area. In Australia, a humpback (*Megaptera novaeangliae*) and southern right whale (*Eubalaena australis*) have travelled straight through finfish farm structures, destroying the cages and/or entangling themselves while following their traditional migration route (Kemper & Gibbs 2001; Kemper *et al.* 2003). Field and captive studies have found that some dolphin and porpoise species are reluctant to swim through vertical structures or those with ropes, usually preferring to swim under rather than crossing over a submerged structure (Kastelein *et al.* 1995; Watson-Capps & Mann 2005; Heinrich 2006). A recent study in Chile did not observe



any of the three local, small cetaceans (*Cephalorhynchus eutropia*, *Lagenorhynchus australis* and *Phocoena spinipennis*) within nearby shellfish farms (Heinrich 2006). The author suggested that farms were excluding dolphins from large and perhaps former habitats of the Bahia Yaldad, despite some individuals travelling within 100 m of farm structures and even feeding on school fish near farm boundaries. Given that some marine mammal exclusion effects occur, the question remains as to the significance of this effect for particular species.

Pinnipeds (*i.e.* seals and sea lions) are perhaps the one marine mammal species that will not be excluded from habitats by mussel farming. In fact, in the Marlborough Sounds, seals are abundant in the vicinity of salmon farm sites, which they have used as a source of food and as haul-out areas previously (especially where seal exclusion nets are not in place or are compromised, Forrest *et al.* 2007).

#### Underwater noise

Associated closely with habitat exclusion is habitat degradation in the form of underwater noise disturbance. Underwater noise in the oceans has recently made headlines as a fairly widespread, yet largely unknown problem for marine mammals, particularly the larger whale species (Nowacek *et al.* 2007; Weilgart 2007; Wright 2008). The amount of low-frequency noise (10-300 Hz) produced by ship and boats in the oceans, and which corresponds to the same ranges at which most baleen whales communicate, is doubling every decade (Thomson & Richardson 1995; Wright 2008). The level and persistence of any associated underwater noises with mussel farming may be minimal relative to other underwater noise sources, such as commercial vessels, but will vary according to farm features (*e.g.* type, size), habitat characteristics (*e.g.* location, depth, type of bottom sediments, shape of coastline) and compounding factors, such as the number of farms and/or other noise sources in nearby regions.

Currently, no New Zealand or overseas studies have specifically analysed noise production in association with aquaculture and marine mammals. As Wursig & Gailey (2002) pointed out, "...the amounts of habitat degradation and reduction of communication capabilities of marine mammals near shellfish farms are simply unstudied." Some overseas studies have demonstrated decreasing use or complete abandonment of habitats by gray and humpback whale populations in which shipping and boat traffic have increased (Gard 1974; Herman 1979; Bryant *et al.* 1984; Glockner-Ferrari & Ferrari 1990). In general, most odontocete (*i.e.* toothed whales and dolphins) and pinniped species demonstrate few avoidance behaviours and considerable tolerance of most underwater noises with a few exceptions (Richardson 1995). In fact, the curiosity and temporary attraction of dolphins to boat noise will be familiar to most recreational or commercial vessel users, and has been recognised in the literature (Carwardine 1995; Dawson *et al.* 2000).

#### Entanglement

Mussel farming structures can occupy a large portion of the water column, as discussed above, effectively creating a three-dimensional obstacle that resident marine mammals have to navigate around (Würsig & Gailey 2002; Markowitz *et al.* 2004). In addition, many species of

marine mammals are known for their curious nature and are often attracted to novel objects, such as floating debris and/or lines. To date there is only one (disputed) report of marine mammal entanglement in relation to mussel aquaculture in New Zealand that we discuss below. The risk of entanglement in relation to mussel farms is discussed within the context of other forms of aquaculture or fishing activity both from within New Zealand waters and overseas.

Several incidents of baleen whale entanglement in lines from shellfish fisheries and/or aquaculture have occurred in Australia and New Zealand. Numerous humpback and southern right whales off Australia have been found entangled and dead in fishing gear as simple as a single craypot line (Allen & Bejder 2003; data obtained from New South Wales, Queensland and Western Australia Department of Conservation and Land Management, Kemper *et al.* 2008). Humpback whale entanglement in craypots has also been reported in New Zealand waters (Suisted & Neale 2004) as well as a Bryde's whale (*Balaenoptera brydei*) found entangled and dead in a spat collection rope from a mussel farm at Great Barrier Island (Seafood New Zealand 1996). In the latter case, the spat line was lodged tightly through the base o the animal's mouth indicating a high level of force would have been necessary to cause the entanglement. However, whether the whale was alive at the time of entanglement has been disputed by industry representatives as no official necropsy was performed.

To our knowledge there are no known records of dolphin or pinniped entanglement in mussel aquaculture lines in New Zealand. Three separate incidents of Hector's dolphin entanglement in New Zealand have involved craypot lines (DoC 2008). Similarly, while some incidences of dolphin entanglement in lines have been reported from overseas, none have been reported for mussel aquaculture. Along the east coast of the United States, for example, bottlenose dolphin (*Tursiops truncatus*) have entangled in crab trap lines (Noke & O'Dell 2002). But the relevance of these types of entanglements to mussel farming is questionable.

In general, it appears to be loose, thin lines that pose the greatest entanglement threat to whales and dolphins as evident by reports in both New Zealand and overseas. As such, potential entanglement risks at New Zealand mussel farms are likely to be low, since backbone lines are under considerable tension. Of secondary and more minor concern are long-line crop ropes hung in continuous loops. Although not documented, this looping configuration has the potential to entangle larger whale species using inshore waters (*e.g.* southern right, Bryde's and humpback whales) due to the animals' girth, long pectoral fins and horizontal orientation of their tail flukes (fluke width can be up to half of total body length; for an average right whale, fluke width would be 9 m). Only one incident of a humpback calf in western Australia becoming entangled in a crop line in its mouth has been reported (Coughran 2005). The calf, after having picked up the line in its mouth, panicked and rolled with the line and had to be cut free from its connection to the farm's anchor due to the strong tension on the 20 mm line. Potential entanglement risks in New Zealand mussel farms are likely to be low, based on the lack of loose lines. However, based on overseas evidence, the risk of this occurring would obviously increase if a farm were situated in a historical migratory path.



#### **Indirect effects**

The potential for wider, more indirect ecosystem effects on marine mammals due to mussel aquaculture also include food-web interactions (Black 2001; Kaiser 2001; Würsig & Gailey 2002; Kemper *et al.* 2003), biotoxin and pathogen (disease) outbreaks (Geraci *et al.* 1999; Kaiser 2001), and antibiotic use (Buschmann *et al.* 1996; Kaiser 2001). While these potential indirect interactions between marine mammals and shellfish aquaculture have been considered in the literature (Würsig & Gailey 2002; Kemper *et al.* 2003), no actual research on any indirect effect has yet been documented.

# 2.5.5. Biosecurity risks and biofouling pests

#### Background

Human activities in New Zealand coastal areas are a significant mechanism for the dispersal of marine pests, particularly the movements of recreational and commercial vessels, and aquaculture activities (Dodgshun et al. 2007). Internationally, the role of aquaculture in the spread of fouling pests has long been recognised (Perez et al. 1981; Bourdouresque et al. 1985; Wasson et al. 2001; Leppäkoski et al. 2002; Hewitt et al. 2004). Awareness of this issue in New Zealand was largely precipitated in the late 1990s by concerns regarding the humanmediated spread and ecological effects of the Asian kelp Undaria pinnatifida (Sinner et al. 2000). Around this time, fouling also became recognised as a significant threat to the mussel industry when a population explosion of the sea squirt Ciona intestinalis resulted in mussel crop losses in parts of the Marlborough Sounds. Subsequently, other fouling pests have emerged whose potential for adverse effects on the mussel industry and the wider ecosystem have been recognised, such as the sea squirts Styela clava and Didemnum vexillum (Figure 13; Coutts & Forrest 2007; Gust et al. 2007). While many of these pest organisms have reached problematical densities only on mussel farms and other artificial structures in New Zealand, overseas evidence also reveals their potential to be highly invasive in natural habitats (e.g. Didemnum; Bullard et al. 2007).

# Spread of fouling pests via aquaculture

The propensity for aquaculture activities to spread fouling pests arises from the fact that suspended cultivation methods, and their associated structures and materials (*e.g.* ropes, floats pontoons), provide ideal habitats that allow such organisms to proliferate at high densities (Clapin & Evans 1995; Floc'h *et al.* 1996; Carver *et al.* 2003; Lane & Willemsen 2004; Coutts & Forrest 2007). From a biosecurity perspective, and for mussel farming in particular, ecological risks arise because the infested farm or other structures act as a 'reservoir' for the further spread of the pest.

At local scales (*e.g.* within bays), spread from infested reservoirs is facilitated by microscopic life-stages (*e.g.* seaweed spores or animal larvae) that are released by adult populations and drift with water currents as part of the seston. For some species dispersal can also occur via the drift of reproductively viable fragments (Forrest *et al.* 2000; Bullard *et al.* 2007). These types of processes can lead to the establishment of the pest on adjacent structures such as other

marine farms, jetties and vessel moorings. In this way such structures can act as 'stepping stones' for the spread of pest species (Bulleri & Airoldi 2005; Forrest *et al.* 2008).

For many fouling organisms, however, natural dispersal is limited, and spread across large areas or between regions occurs via inadvertent transport with aquaculture and other human activities (*e.g.* vessel movements). For example, infested structures deployed at a mussel farm (*e.g.* ropes, floats), farm vessels, or seed stock, may be transferred to other localities as part of routine operations. There is a high likelihood that associated fouling organisms will survive where such transfers occur without the application of measures to reduce biosecurity risks (Forrest *et al.* 2007). In recognition of this, the mussel industry in New Zealand, along with other aquaculture sectors, has been proactive in the development of biosecurity management measures, as described below.



**Figure 13.** Fouling on mussel farms in the Marlborough Sounds; a. The sea squirt *Didemnum vexillum* and kelp *Undaria pinnatifida* on mussel crop lines; b. general fouling of mussel crop lines; c. mussel farm float covered in the sea squirt *Ciona intestinalis* (B Forrest, Cawthron).

#### Existing and potential biosecurity risks from mussel farming in New Zealand

In the absence of effective management measures, the risk of inter-regional spread of pest organisms by mussel farm activities in New Zealand can be relatively high in certain circumstances. While a number of farm-related mechanisms have been described (Forrest & Blakemore 2002), transfers of seed stock within and between mussel farming regions are of particular significance, and have resulted in the spread of a number of high profile pests to date in New Zealand (*e.g.* the kelp *Undaria* and sea squirt *Didemnum*). The main pathways of mussel seed stock transfer (along with oyster industry and other aquaculture transfers) were described by Forrest & Blakemore (2002) and the two main transfer pathways for seed stock are shown in Figure 14. As noted in Section 2.6, approximately 80% of industry seed-stock needs are met by transfers of 'Kaitaia spat', with other increasingly common pathways being

movements of spat from other regions (especially Tasman and Golden Bays) and 'seed mussels' (referring to mussels that are typically 20-60 mm in length) between growing regions. In all cases, biosecurity risks have emerged as described below.

In May 2000, a bloom of the planktonic microalga *Gymnodinium catenatum* was detected off New Zealand's northwest coastline, in an area that encompassed the source of Kaitaia spat (MacKenzie & Beauchamp 2000). *Gymnodinium catenatum* produces biotoxins that result in paralytic shellfish poisoning in humans that eat infected shellfish, and has been responsible for closures of shellfish aquaculture areas worldwide (Rhodes *et al.* 2001). Hence, the subsequent detection of high densities of *Gymnodinium* cysts in Kaitaia spat supplies led to a voluntary industry ban on spat movements to all aquaculture regions, and the development of treatments to minimise cyst densities within infected material so that inter-regional transfers could continue (Taylor 2000). The main pathway targeted was spat transfer to the Marlborough Sounds; the fact that *Gymnodinium* never fully established in the Sounds may in part be attributable to these management efforts (Forrest *et al.* 2008). Kaitaia spat transfer protocols are now incorporated into a voluntary industry code of practice developed by the New Zealand Mussel Industry Council (NZMIC).

While the *Gymnodinium* issue has died down, the transfer of fouling pests with seed mussel movements has remained a difficult and ongoing management issue for the mussel industry. The problem is that when seed mussels are transferred by road between growing regions, associated fouling organisms can sometimes survive and be transferred as well (Forrest & Blakemore 2006). A Code of Practice produced by NZMIC in 2001 sought to minimise this risk. The Code identifies three geographic mussel farming zones<sup>5</sup> and requires that seed mussels moved between these zones be declumped, thoroughly washed, transferred as single seed (*i.e.* not attached to each other by their byssus), and visually free of several target species (native blue mussels, and non-indigenous *Ciona intestinalis* and *Undaria pinnatifida*). This code will undoubtedly reduce the transfer of the target species between the three zones; however, microscopic life-stages and reproductively viable fragments can survive the declumping and washing process (Forrest & Blakemore 2003, 2006). For example, inadvertent transfers of the sea squirt *Didemnum vexillum* occurred among growing areas within the Marlborough Sounds despite application of the declumping and washing process, in some cases leading to the application of more stringent treatments.

There has been considerable research in to secondary treatment tools to completely eliminate pest risks associated with seed mussel transfer (Forrest & Blakemore 2006; Denny & Hopkins 2007; Forrest *et al.* 2007). While a number of promising methods have been advanced (*e.g.* heat, eco-friendly chemicals) or are used at an operational scale to some extent (*e.g.* freshwater), the mussel industry is still in need of a widely applicable and 'bullet proof' method that is operationally feasible and affordable. The key challenge is to find a method that will kill target pests without adversely affecting the health of the mussels. The present

<sup>&</sup>lt;sup>5</sup> The three zones are: northern New Zealand (north of Mahia Peninsula including the Firth of Thames and Coromandel); southern New Zealand (south of Kaikoura); and a central zone between these two (which includes the Marlborough Sounds and Golden/Tasman Bays).

focus of industry efforts has been on developing methods that are effective against the sea squirt *Didemnum vexillum*. This species has been subject to considerable industry-led management efforts in the Marlborough Sounds, due to concerns regarding its potential fouling effects on the mussel industry. While eradication has failed, the industry still wishes to contain the spread of *Didemnum* and prevent its inadvertent transfer among New Zealand's main spat catching and growing regions. Emerging evidence that *Didemnum* in New Zealand has the potential to colonise high-value 'structured' ecological habitats (*e.g.* horse mussels, hydroid trees, kelp beds, black coral trees), suggests that the benefits of effective management will extend beyond the aquaculture industry alone (B. Forrest, pers. obs.).

The development of new mussel farm operations, especially in regions where no marine farming exists (Figure 14), raises the likelihood that biosecurity risks will arise. In general terms, risks will be most significant when: (i) pest organisms are spread by mussel farm activities into regions or habitats that are optimal for their establishment and where they do not already exist; and (ii) mussel farming activities are the primary mechanism for the spread of the pests. If a pest organism is already present in the new habitat, or is likely to spread there regardless of mussel aquaculture activities, for example via natural dispersal or via non-aquaculture vectors (*e.g.* recreational vessels), then the incremental risk posed by mussel farm operations may be negligible. Determination of such risks is situation-specific and must be evaluated on a case-by-case basis. Provided there is knowledge of the biological attributes of pest organisms (*e.g.* natural dispersal capacity and habitat requirements) and human-mediated pathways of spread (Dodgshun *et al.* 2007), various assessment procedures can be used to assist with identification of relative risks and the extent to which they can be managed (Forrest *et al.* 2006).





**Figure 14.** Existing (A-H) mussel farming regions showing the main pathways of equipment/vessels, Kaitaia mussel spat and seed-mussels around New Zealand. Future pathways may also arise as proposed new regions develop (Figure modified from Dodgshun *et al.* 2007).

#### 2.5.6. Disease

#### Overview

The risk of transmission of pathogens or parasites from cultured to wild *P. canaliculus* to other species can be considered minimal at present. With the exception of the protozoan parasite apicomplexan X, (APX), all other diseases reported in cultured mussels usually have lower prevalence's and intensities than wild mussels (S Webb, pers. obs.). Furthermore, no mussels in New Zealand have been reported with any pathogens appearing on the *Office International des Epizooties* (OIE) list of important diseases (Webb 2007). The effects of disease on the farmed mussels themselves are of importance with regard to farm management and can be economically significant. A detailed description of diseases known to affect non-finfish aquaculture species both in New Zealand and overseas is provided in Appendix 2.

# Disease in New Zealand Greenshell<sup>TM</sup> mussels

Greenshell<sup>TM</sup> mussels are not highly prone to disease. Past studies have found no diseaseassociated mortalities in Greenshell<sup>TM</sup> mussels (Hine 1989) or the presence of potentially serious pathogens within the mussels (Hine 1996). A recent review on mytilids with particular emphasis on *P. canaliculus* (Webb 2007) indicates that there have been no particularly destructive diseases of mussel species identified in New Zealand, with the exception of a digestive viral disease. Digestive viral disease (digestive epithelial virosis) was first noted by Jones *et al.* (1996) who reported mortalities in cultured Greenshell<sup>TM</sup> mussels in the outer Marlborough Sounds of which the majority were associated with virus-like particles and digestive tubule damage. The condition also affects scallops and clams in New Zealand and other bivalve molluscs elsewhere. Viruses producing similar digestive tissue effects on bivalve molluscs have been reported in Australia, Scotland, Denmark, and elsewhere (Bower 2001).

Another pathogen that poses potential environmental risk is the parasite APX, which is reported from New Zealand only (Diggles *et al.* 2002; Hine 2002b) and has been found in mussels from the Marlborough Sounds and also occurs commonly in dredge oysters *O. chilensis* (also known as flat oyster) from all around the coast (Diggles *et al.* 2002; Hine 2002b). In oysters, APX can cause a significant condition referred to as coccidiosis (Hine & Jones 1994); however, its effect on mussels is less noteworthy. Cultured Greenshell<sup>TM</sup> mussels appear to present no major threat to wild molluscs, as wild Greenshell<sup>TM</sup> stocks can harbour all known pathogens with the exception of APX. Since APX is also found in dredge oysters, however, there would remain a reservoir of infection even in the absence of Greenshell<sup>TM</sup> mussel culture.

The threat to wild mussels and other bivalve species from farmed mussels carrying indigenous diseases/parasites is therefore presumably low. Known pathogens in New Zealand occur in a range of other wild bivalve species, often at a greater prevalence and intensity than in cultured mussels. Farmed mussels could pose a threat if they were vehicles for introduction of an exotic disease (see discussion on *Marteilia* in Appendix 3). This is a possibility only if *P. canaliculus* is susceptible and if appropriate intermediate hosts (if required) are available. The possibility that potential intermediate hosts could be part of the suite of fouling organisms should not be overlooked, both in life cycle studies and as possible control measures.

#### Effects of disease documented overseas

There is evidence from overseas that mussels can harbour pathogens that are transferable to fish. The aquabirnavirus infectious pancreatic necrosis virus (IPNV) has been detected in *Mytilus edulis* (VPS 2000). It is a common virus of salmonids and is also a suspected clam pathogen in Taiwan. *M. galloprovincialis* was identified as a reservoir host for infections of the aquatic birnavirus (ABV) in the Japanese flounder *Paralichthys olivaceou* (Kitamura *et al.* 2007). This internationally significant disease of world-wide distribution is reported in healthy King salmon (*Oncorhynchus tshawytscha*) returning from the sea on the east coast of South Island, New Zealand (Diggles *et al.* 2002). Although not detected in New Zealand mussels, the possibility of *P. canaliculus* harbouring this virus, at least temporarily, finds support in the reports by Lewis *et al.* (1986) and Greening *et al.* (2001) where polioviruses and enteroviruses have been shown to persist in *P. canaliculus* after experimental exposure. Caution is clearly required in polyculture, as mytilids might harbour viruses with consequent threat to susceptible fish.



# 2.5.7. Genetic interactions with wild populations

#### Overview

As described in detail in Appendix 3, there is potential for aquaculture to affect genetic profiles of wild populations of the same species. The problem stems mainly from the act of shifting significant numbers of individuals of a single species and establishing them elsewhere. Potential for altering genetic profiles of wild populations is largely determined by the pre-existing level of genetic structuring within that species. The level of genetic structuring is essentially a product of evolution; reflecting the spatial distribution of the species and associated variations in environmental conditions, as well as the extent of connectivity (gene flow) within and between populations. The conditioning of animals to local conditions acts on small and large spatial scales (hundreds of metres to hundreds of kilometres) by affecting fitness of individuals and populations (*i.e.* through natural selection).

Thus, pertinent to this assessment is the fact that *P. canaliculus* is endemic to, and found throughout, the latitudinal extent of New Zealand. Approximately ~80% of the spat (small mussels) used by industry are collected on drift *Sargussum* seaweed on Ninety Mile Beach in the far north (approx 35° S) which is then trucked to three main aquaculture centres – the east coast of the Coromandel Peninsula (approx 37° S), the Marlborough Sounds (approx 41° S), and Stewart Island (approx 47° S) (Hickman 1983). The remaining ~20% is sourced from spat collection devices in Golden Bay and a few other small sites in the Marlborough Sounds and Aotea Harbour. It is by far the largest aquaculture industry in New Zealand and involves the relocation of very large numbers of organisms.

# Genetic connectivity and variation among populations of P. canaliculus

There are well documented cases of self-recruitment (reviewed by Swearer *et al.* 2002) and increasing evidence that this phenomenon is not as rare as first thought (Wood & Gardner 2007). However, in the context of the extent and patterns of larval connectivity (= gene flow) among *P. canaliculus* populations, it is safe to say that self-recruitment is not likely to be an important consideration in the present situation. Because of the vast numbers of offspring produced by a broadcast spawning species such as *P. canaliculus*, and because of the highly dispersive nature of the larval stage (an evolutionary adaptation to promote gene flow among populations and the colonisation of new environments), it has been reasonably assumed that reproductive connectivity is high among populations.

Studies have identified different models of genetic structuring within populations of *P*. *canaliculus* (a putative north-south split, Smith 1988; an isolation-by-distance model, Gardner *et al.* 1996a; and a panmixia model, Apte & Gardner 2001 – see Appendix 3 for details) and have consistently demonstrated high levels of genetic variation within the species. The most recent studies, which employed modern molecular approaches (Apte & Gardner 2002; Star *et al.* 2003), made the following significant findings.

A pronounced genetic discontinuity exists among Greenshell<sup>TM</sup> mussel populations at ~42°S, such that a northern group can be clearly recognised and differentiated from a southern group of mussels. Populations found within the northern group tend to be more genetically similar to each other than those within the southern group. Such genetic-



based stock differences are likely to be associated with fitness differences and can be used as a signal to track the movement of stocks from Kaitaia (in the northern group) to Stewart Island (in the southern group).

- 2. At Stewart Island, the wild (natural) mussel population at Horse Shoe Bay, which is <20 km from the Big Glory Bay aquaculture site, shows intermediate affiliation with the northern and southern groups, when it should only exhibit affinity to the southern group. This intermediate status is clear evidence of the introgression of northern genes into this wild southern population. The geographical extent of this event is unknown. This observation is important as introgression has potentially important ramifications for 'fitness' of the resulting hybrids (see Appendix 3).
- 3. The genetically intermediate identity of the Horse Shoe Bay population between the northern and southern stocks strongly suggests that not only have northern genes introgressed into the southern population, but that hybrids and individuals of mixed ancestry (*i.e.* possible backcrosses) are reproductively active.

In terms of the likely fitness consequences of interbreeding of northern and southern Greenshell<sup>TM</sup> mussels, there is very little published information on which to base predictions about relative hybrid fitness, and there is no clear indication from the literature about what to expect. There are however a number of possible scenarios that can be inferred from experiences with other species, which vary in relevance and have been discussed in some detail in Appendix 3. The conceivable consequences of adversely affecting the fitness of wild populations are significant in both magnitude and permanence, and warrant some further consideration. However the susceptibility of *P. canaliculus* to genetic issues is mitigated by the following factors:

- The connectivity among *P. canaliculus* populations is relatively high and the genetic structuring within regions is relatively low, although genetic differences between regions are high.
- The pre-existing level of transfer is high, and 'fitness' issues are yet to be observed (although they have not yet been rigorously examined).
- The industry is presently based on wild-sourced progeny that have genetic diversity similar to mussels from two of the three main aquaculture areas.

Based on these factors, and as far as it is possible to predict with the available information, the continued transfer of wild-sourced mussels within and between the northern and southern groups is unlikely to adversely affect fitness of wild stocks in the future. However, this conclusion does not hold if the mussel industry were to increase its dependence on hatchery-supplied spat, particularly with the development of selectively bred animals. This would present new implications that would need to be carefully considered and likely require the development and implementation of genetic management protocols.



# 2.6. Environmental effects of Greenshell<sup>™</sup> mussel spat catching

# 2.6.1. Background

The word 'spat' is used to describe young mussels from the time that they have settled out of the water column and metamorphosed to resemble the adult form, until they are large enough to be transferred to a farm for ongrowing (*i.e.* from ~1 to 10 mm length). Spat are typically 'caught' by placing specially designed frames wrapped in hairy rope, or long dropper ropes in the water column, to provide surfaces that encourage larval settlement. Farmers aim to synchronise gear placement with the time mussel larvae are most abundant in the water (usually early to late summer), and then remove the gear as soon as the mussels achieve transfer size. This task is made difficult because natural spawning events, and the subsequent larval development and settlement processes are inherently highly variable and hard to predict. Therefore in some instances, spat collecting gear may be kept in the water throughout the year. Year-round spat catching activities are generally carried out in close association with mussel out-growing at the same or nearby sites.

There are occasions when the lines are colonised (or 'over-settled') by mussels or non-target species (*e.g.* scallops). Over settlement can result in large-scale spat drop-off to the seabed, particularly when the collectors are subjected to storm disturbance. Thus, there is potential for enhanced recruitment of adult forms of the colonising species on the seabed beneath the structures. In Tasman and Golden Bays, this applies to both mussels and scallops; the former being the target species and the latter an important commercial fishery species. An unnaturally high abundance of either or both species has the potential to create an ecological imbalance with possible ramifications for the surrounding scallop fishery.

Approximately 20-25% of the mussel industry presently sources its stock from designated spat catching farms (G Coates, New Zealand Marine Farming Association, pers. comm.). The remaining stocks are sourced from the wild (largely from Northland beach-cast macroalgae) and transported to growing areas in other parts of the country (see discussion in Section 2.5.5). Until recently, the vast majority of spat farms have been situated within Golden and Tasman bays at the top of the South Island. Small spat catching farms also exist elsewhere, including Coromandel, the Marlborough Sounds and Aotea Harbour. The number and extent of areas designated for spat catching is due to expand considerably with the recent approval of several new sites, also situated in Golden and Tasman bays. Some of these new areas are permitted for permanent occupation and, as such, will not have the confounding disturbances from dredging in-between seasons and will provide a better indication as to the specific effects of spat catching. However, some of these sites are also permitted for mussel grow-out and may become dual purpose (*i.e.* periodically utilised for mussel grow-out). In such cases, the specific effects will be difficult to discern from those of the grow-out phase. The feasibility of spat catching on large offshore sites has also been examined as part of a FRST-funded research project (CAWX0302). The management of these new sites may vary slightly from those considered in the present assessment, primarily because they are permitted for permanent occupation. However, their use during the spat catching season should be roughly analogous

in terms of period, duration and catching methods. As such, findings of the environmental monitoring that has been conducted so far should be generally applicable.

# 2.6.2. Potential seabed effects

This section summarise potential issues based on environmental data that has been collected during the establishment and operation of New Zealand's largest dedicated spat catching farms located in Tasman and Golden Bays. The data was collected as part of an Environmental Adaptive Management Plan (EAMP) prepared under contract to the developers (Hopkins 2005). To date, spat catching in these areas has been a temporary activity with all structures being removed between seasons. The main environmental consequence of periodic spat catching was that the sites were targeted for dredging between seasons due to their propensity to promote settlement of juvenile scallops, which were subsequently used for enhancement of the Nelson Bays scallop fishery. It was therefore difficult to distinguish the effects of spat catching facilities from those of mechanical disturbance by dredges. Included in the environmental monitoring program were: regular density assessments of scallops, mussels and predators (*e.g.* starfish), density-dependant pathological examinations, and effects to infauna and sediment characteristics. The results are summarised from various Cawthron client reports prepared for the industry (Hokpins *et al.* 2005; Keeley 2007; Keeley & Forrest 2008; and numerous monthly letter reports).

#### Alteration of benthic physical and biological properties

Spat catching involves culturing high densities of filter feeding bivalves that produce waste materials and therefore have the potential to cause analogous depositional and enrichment effects as with mussel grow-out (see Section 2.3). The scope for enrichment effects is somewhat mitigated by two factors. Firstly, spat catching is generally a seasonal activity with lines removed for at least six months of the year. Secondly, although not well described in the literature, the energetic requirements of very small mussels (*i.e.* spat: 5-10 mm) are likely to be proportionate to their body mass. Since the relationship between length and tissue mass is exponential, the feeding requirements of spat are likely to be correspondingly low. In one of the few studies that considers mussel size in relation to feeding and excretion rates (James *et al.* 2001), it was apparent that a non-linear (power) relationship existed. Extrapolating backwards to a *c.* 10 mm mussel would suggest that clearance (litres filtered/mussel/hr) and excretion rates would be very low indeed. Hence, the potential for deposition- and enrichment-related effects beneath spat catching sites that hold comparatively low biomasses of shellfish would presumably be less than that of grow-out operations.

Monitoring results collected to date are consistent with this expectation. After four years of operation, the physical and biological properties of the sediments beneath seasonal spat catching sites had not changed appreciably (Keeley & Forrest 2008). The small changes that were observed in the physical composition of the sediment were more likely attributable to mechanical disturbance from dredging between seasons than deposition from spat catching activities.



#### Accumulation of shellfish spat on the seabed

There is a potential for the densities of shellfish to increase beneath spat collecting structures to the point where they cause ecological imbalances and/or facilitate the spread of disease. Elevated densities were observed in Tasman and Golden Bays, but this process was often constrained by environmental controls (Hopkins et al. 2005; Keeley & Forrest 2008). For example natural storm-related sedimentation events in river plume-affected regions can smother juvenile shellfish, and regular tidal resuspension from the seabed can cause chronically elevated near-bottom suspended sediment concentrations that interfere with shellfish feeding (Gillespie & Rhodes 2006). Consequently, elevated shellfish densities have not been seen to cause undue ecological stress at these locations. Nonetheless shellfish density monitoring was continued in order to identify management practices that would minimise spat drop-off and enable density reduction by dredging, if deemed necessary (Hopkins et al. 2005). Although site clean-up through dredging has been required when shellfish trigger densities have been exceeded, this is not an ideal outcome as it disturbs the seabed and has its own range of environmental issues (Thrush et al. 1993; Dolmer 2002). Dredging is only justified if it is clear that the resulting densities are creating ecological imbalances, and this has not been observed to date.

#### 2.6.3. Water column effects

Although spat are smaller than adult mussels, there is still a potential for the same suite of water column issues that are described for mussel out-growing in Section 2.4. But as with the depositional effects, the effects are mitigated in juvenile bivalves by the lower rates of filtration and subsequently excretion (James *et al.* 2001). Chl *a* spatial surveys conducted as part of the Tasman and Golden Bay EAMP failed to identify any depletion shadows that would be consistent with localised food depletion. More detailed surveys are planned to confirm these observations; however, chl *a* depletion and associated carrying capacity issues around spat farms are expected to be negligible.

# 2.6.4. Wider ecological effects

Wider ecological effects of spat collecting (*i.e.* in relation to fish, mammals, biosecurity, disease *etc.*) will be largely similar to those described for the grow-out phase of mussel cultivation (see Section 2.5). One possible exception is that spat farms may be more likely to pose entanglement risk to marine mammals based on their location in relation to important feeding or birthing habitats and migratory routes (see Section 2.5.4). The spread of disease and poor health from over-crowding were identified above as potential issues associated with spat catching activities. However, comprehensive pathological examinations of scallop and mussel spat at Tasman and Golden Bay sites where shellfish densities of >100/m<sup>2</sup> were achieved revealed no diseases, viruses or parasites considered to be indicative of poor health (Webb & Govier 2007, also summarised in Keeley 2007). While this was an encouraging result, it does not preclude the possibility that disease outbreaks could occur in the future or that elevated shellfish densities could, in turn, increase susceptibility and/or facilitate spread of



disease. Such epidemics can occur in naturally sustained fishery stocks (*e.g.* Coromandel scallops, Diggles 1999) and this needs to be taken into consideration if an outbreak occurs. See Section 2.5.6 and Appendix 2 for more detailed information on the pathology of mussels and other bivalves.

#### Aggregation of invertebrate predators

A possible wider ecological effect arising from shellfish drop-off and accumulation, is the potential response of mobile predatory epifauna; particularly those that effectively prey on juvenile shellfish. In Tasman and Golden Bays, the epifauna is generally sparse, and significant predatory species are limited to starfish (*Coscinasterias calamaria*), octopus, hermit crabs (*Pagurus* sp.) and predatory whelks. An elevated abundance of one or more of these species could potentially affect naturally occurring prey species in the vicinity of the farm, particularly between seasons when the lines are removed. The densities of these predator species were monitored over five consecutive years in conjunction with the spat density monitoring in Tasman and Golden Bays (Keeley 2007). The results indicated that effects to mobile predators were negligible over the monitoring period. However, if drop-off is not managed effectively and spat densities build up, then it is conceivable that predator abundances will respond accordingly.



# 3. ECOLOGICAL EFFECTS OF INTERTIDAL PACIFIC OYSTER CULTIVATION

# 3.1. Introduction

Intertidal oyster cultivation is one of the most significant aquaculture industries world-wide (FAO 2006). While the global industry is based on a range of species, Pacific oysters (*Crassostrea gigas*) are by far the most dominant, having been spread either deliberately or inadvertently (*e.g.* via shipping) to many countries (Kaiser *et al.* 1998). Pacific oysters are endemic to Japan, and were first observed in New Zealand in Northland in 1971 (Dinamani 1971) and were later observed on the South Island (Jenkins & Meredyth-Young 1979). They may have arrived in New Zealand as early as 1958 (Dromgoole & Foster 1983), possibly following inadvertent introduction via shipping mechanisms such as ballast water discharge or hull fouling. Intertidal cultivation of Pacific oysters began in New Zealand on the tidal flats of Northland harbours in the mid-1970s, in favour of an industry at that time based on cultivation of native rock oysters (*Saccostrea glomerata*).

The total area consented for intertidal Pacific oyster farms in New Zealand now exceeds 1000 ha, but not all of the allocated space is in production. As indicated in Section 1, operational farms number more than 200 and occupy approximately 750 ha of estuarine habitat. Most of the intertidal farms are located in the estuaries of northern New Zealand (Figure 1). Key cultivation areas include Whangaroa, Parengarenga, Mahurangi, and Kaipara Harbours, as well as the Coromandel Peninsula, Ohiwa Harbour and Bay of Islands (MFish 2006). The majority of oyster farms in these areas consist of wooden racks (~50 m L x 1 m W x 0.75 m H) in the lower intertidal zone. Sticks to which juvenile oyster spat are attached are laid across these racks and are therefore elevated above the sediment (Figure 15 & 16). To produce single seed (unattached oysters), the oysters are sometimes stripped from sticks and placed in plastic mesh bags or trays. Racks are spaced several metres apart to allow access by farm barges. The spat supply for the northern industry has historically relied on wild-caught spat, mainly from farms in Kaipara Harbour. Increasingly, however, the industry is using selectively bred single-seed oyster spat from a hatchery in Nelson. Last year approximately 20% of the industry spat supply came from this hatchery source.

Owing to a high global demand for Pacific oysters, there is considerable interest in the further development and expansion of this industry in New Zealand. As part of this development, it appears likely that the industry will slowly convert to intertidal long-line culturing using an Australian-designed "BST system" (Figure 16). This system involves enclosing oysters in cages which are suspended from a plastic-coated wire cable strung between posts. The cables can be adjusted in height to provide more control over the farming process (*e.g.* control of oyster growing height, control of biofouling) compared with rigid rack methods. The BST system also provides a more desirable single seed oyster and enables greater mechanisation of the farming operation (Handley & Jeffs 2002; Hay & Lindsay 2003). Most of the main oyster growing areas now have small areas (a few hectares) where the BST method is used. A new oyster farm recently consented for Kaipara Harbour will cover 76 ha when fully developed,

and aims to exclusively use the BST system. There is also increasing interest in the development of subtidal suspended culture methods for Pacific oysters. Subtidal culture is currently undertaken at only a very small scale in New Zealand, or used for oyster fattening (Handley & Jeffs 2002). There are ongoing efforts, however, to develop methods that overcome some of the barriers to successful subtidal culture (*e.g.* excessive fouling), to enable this approach to be more widely used (Hay & Lindsay 2003; Olin Pilcher, Cawthron, pers. comm.).



**Figure 15.** Pacific oyster cultivation areas in northern New Zealand; a. Mahurangi Harbour (R Creese); b. Waikare Inlet (B Howse, Northland Regional Council).



**Figure 16.** Pacific oyster cultivation; a. Elevated intertidal oyster racks in northern New Zealand (B Forrest); b. BST system that encloses oysters within suspended cages (Handley & Jeffs 2002).


## 3.1.1. Scope of this review and information sources

In Section 3 of this report we restrict our review to discussion of ecological effects from intertidal Pacific oyster cultivation, as this represents the mainstay of the New Zealand industry. Effects arising from the future development of subtidal Pacific oyster culture referred to above, and the culture of other oyster species (*e.g.* the flat oyster *O. chilensis*), are discussed separately in Section 4.3.1

To date there has been only one field study of intertidal oyster culture effects in New Zealand (Forrest 1991). That work focused on ecological effects of Pacific oyster racks on seabed habitats in Mahurangi Harbour, and pertinent information from that work has recently been published (Forrest & Creese 2006). A broader range of potential ecological risks was discussed in relation to a proposed Pacific oyster and mussel farm developments in Kaipara Harbour (Gibbs *et al.* 2005; Hewitt *et al.* 2006) and, more recently, a review paper has been produced that provides a synthesis of ecological risks associated with intertidal oyster aquaculture (Forrest *et al.* 2009). The review in turn was an extension of a brief overview of ecological issues associated with Pacific oyster cultivation in Northland, which was produced for Northland Regional Council as part of the Foundation for Research Science and Technology Envirolink advice scheme (Forrest *et al.* 2007). Additional work on the effects of mudworm infestations on cultured oysters has also been conducted (Handley & Bergquist 1997) but wider ecological ramifications were not considered. Based on these information sources, it is apparent that:

- (i) The actual or potential nature and magnitude of effects from oyster farms in New Zealand are similar to that described for comparable forms of intertidal oyster aquaculture in estuaries overseas.
- (ii) The broad interactions of oyster farms with the environment, and the magnitude of effects, are similar to other forms of aquaculture, especially mussel farming (Section 2).

In the sections below we provide an overview of the known or potential effects of Pacific oyster farming in New Zealand within the wider context of other relevant knowledge from New Zealand and overseas. The focus is almost exclusively on the sea-growing stage of intertidal oyster aquaculture. Where appropriate we include reference to on-ground cultivation methods (widely practiced in some countries) and natural or restored oyster reefs to provide insight into the potential ecological roles of intertidal oyster cultivation. Some of the information we present is extracted, often verbatim, from the Forrest *et al.* (2008) review or earlier Envirolink report, and some of the generic information is the same as provided for mussel farming in Section 2. The text is modified as appropriate, however, to reflect the New Zealand oyster farming situation and the particular purpose of this report.

While we recognise that a range of short-term ecological effects may arise as a result of oyster farm construction, and in relation to other aspects of farming operations such as spat collection at farming sites, we make the assumption that the site-specific effects of such activities are similar or less than in the case of the cultivation phase. The only exception is in Section 3.5.5,

where we recognise the potential role of oyster (including spat) movements in the introduction and spread of pest organisms.

# 3.2. Overview of Pacific oyster cultivation issues

The occupation of space by intertidal structures means that oyster cultivation can conflict with a range of other environmental, social and economic values (DeFur & Rader 1995; Simenstad & Fresh 1995; Kaiser *et al.* 1998; Read & Fernandes 2003). However, it is the ecological effects of intertidal oyster farming that have received the most scientific attention internationally, with the literature dominated by papers that describe cultivation effects on sediments and associated biota (Ito & Imai 1955; Kususki 1981; Mariojouls & Sornin 1986; Castel *et al.* 1989; Nugues *et al.* 1996; Spencer *et al.* 1997; De Grave *et al.* 1998; Kaiser *et al.* 1998; Forrest & Creese 2006; Dubois *et al.* 2007). In addition to seabed effects, there are a range of broader ecological issues associated with intertidal oyster aquaculture that are less well recognised or need to be considered in a comparative context (Figure 17). These include the introduction of pests and disease, creation of novel habitat, alteration to water flows and nutrient cycles, and depletion of suspended particulate matter (especially phytoplankton) by oyster crops. Related considerations are the wider ecosystem consequences of such changes, for example implications for fish, seabirds and marine mammals.



**Figure 17.** Schematic of actual and potential ecological effects from intertidal oyster cultivation (modified from Forrest *et al.* 2007).

While the broad range of ecological effects from oyster aquaculture have received some attention in the literature, much of the knowledge-base relates to natural oyster reefs or onground culture methods (Ruesink *et al.* 2005; Powers *et al.* 2007). Furthermore, where the ecological effects of intertidal methods are specifically addressed, the complexity of some of the ecosystem issues and interactions depicted in Figure 17 means they are often considered or reported superficially (Crawford 2003); alternatively where more thorough assessment is undertaken it is usually for specific issues in isolation. The review by Forrest *et al.* (2009) was a more integrated and in-depth assessment in which the relative significance of each issue was considered within the context of the full range of actual or potential ecological effects.

# 3.3. Seabed effects

## 3.3.1. Biodeposition

## Nature and magnitude of depositional effects

Oyster farms act as biological filters that remove suspended particulate matter from the water column as it flows through the culture and processes the material into waste products in the form of faeces and pseudofaeces. These waste products (generally referred to as 'biodeposits') are heavier than their constituent particles, and readily settle on the seabed beneath culture areas (Haven & Morales-Alamo 1966; Kusuki 1981; Mitchell 2006). Since biodeposits are organic-rich and consist of a substantial proportion of fine particles (*i.e.* silt and clay), seabed sediments beneath oyster cultures can become organically enriched and fine-textured relative to surrounding areas, and can have a reduced REDOX potential (Forrest & Creese 2006).

Changes in physico-chemical characteristics stemming from an enrichment of organic material beneath oyster cultures can lead to a displacement of large-bodied macrofauna (*e.g.* heart urchins, brittle stars, large bivalves) and the proliferation of small-bodied disturbance-tolerant 'opportunistic' species (*e.g.* capitellid polychaetes and other marine worms). Localised minor-to-moderate enrichment effects of this nature have been described (to varying degrees) beneath intertidal oyster farms in Mahurangi Harbour (Forrest 1991; Forrest & Creese 2006) and in numerous studies overseas (Kususki 1981; Mariojouls & Sornin 1986; Nugues *et al.* 1996; Spencer *et al.* 1997; De Grave *et al.* 1998; Kaiser *et al.* 1998; Forrest & Creese 2006; Dubois *et al.* 2007). Castel *et al.* (1989) also described an increased meiofaunal density and biomass beneath oyster trestles in France.

Without exception, it is apparent that direct biodeposition effects associated with oyster cultivation are highly localised to farmed areas (extending tens of metres or less from structures in Mahurangi Harbour) and greater directly beneath racks than between them (Forrest & Creese 2006). The magnitude of biodeposition effects appears comparable for that described for subtidal mussel culture in New Zealand (Section 2; Kaspar *et al.* 1985) but relatively minor by comparison with that described for some mussel culture areas overseas (Mattsson & Lindén 1983; Grant *et al.* 1998) and the suspended subtidal culture of fish (Brown *et al.* 1987; Karakassis *et al.* 2000; Forrest *et al.* 2007). Extreme enrichment effects in relation to oyster farming have been described only for suspended culture systems in Japan,

and had been attributed to repeated culturing and over-stocking (Ito & Imai 1955; Kusuki 1981).

#### Factors affecting the magnitude and spatial extent of seabed effects

The magnitude of effects from biodeposition will depend primarily on oyster stocking density and biomass in relation to the flushing characteristics of the environment (Pearson & Black 2001). Additionally, the level of biodeposition for a given stocking density, and the assimilative capacity of the environment, may vary seasonally (Kusuki 1981; Souchu et al. 2001; Mitchell 2006). To our knowledge, the relative role of these different attributes has not been quantified for oyster farms. As with other forms of aquaculture, the capacity of the environment to assimilate and disperse farm wastes will mainly depend on water current velocity and wave action (Souchu et al. 2001), as these factors control the size and concentration of the depositional 'footprint'. Increased flushing from currents and waves will reduce biodeposit accumulation and increase oxygen delivery to the sediments, thus allowing for greater assimilation of farm wastes (Findlay & Watling 1997; Mitchell 2006). Negligible enrichment effects from intertidal oyster farms in Tasmania have been attributed to a combination of low stocking densities and adequate flushing (Crawford 2003; Crawford et al. 2003; Mitchell 2006). Similarly, experience with fish farming shows that well-flushed sites have depositional footprints that are less intense (but more widely dispersed) than shallow, poorly flushed sites (Pearson & Black 2001).

### 3.3.2. Accumulation of shell litter, debris and associated organisms

The accumulation of live oysters, shell litter and farm debris (*e.g.* oyster growing sticks), and fouling or epibenthic organisms beneath growing racks can be the most visible effects of oyster farms during low tide. Oyster shell and debris is evident, for example, at Mahurangi Harbour and Waikare Inlet oyster farms as in Figure 18. The extent of drop-off to the seabed is likely to depend on the type of cultivation system (*e.g.* stick culture is likely to deposit more debris than basket culture) and may be exacerbated periodically during harvesting. The degree of fouling accumulation will depend on the degree to which structures become fouled, and patterns of natural drop-off or active defouling by farm personnel. Subsequent effects to benthic community composition, for example aggregation of carnivorous and deposit feeding species in response to the food supply (*e.g.* sea stars) and competition between deposited shellfish and benthic filter-feeders, are indicated for other forms of bivalve aquaculture (Smith & Shackley 2004; Hartstein & Rowden 2008) and conceivably occur in the case of intertidal oyster culture. Excessive deposition and decay of fouling biomass may also exacerbate the organic enrichment described above, although such effects would likely be patchy beneath cultivation areas.

Hard surfaces such as live and dead oysters, calcareous debris (*e.g.* bivalve shells, serpulid polychaete tubes) and farm materials potentially provide novel habitats for fouling organisms and associated mobile biota, which would otherwise not occur (or be at reduced densities) in the absence of oyster growing. Such effects have been widely documented overseas in the case of on-ground shellfish culture (Dumbauld *et al.* 2001; Hosack *et al.* 2006; Powers *et al.* 



2007) and oyster reefs (Peterson *et al.* 2003; Escapa *et al.* 2004; Ruesink *et al.* 2005; Coen *et al.* 2007). For example, the structured habitats provided by oyster reefs can support a diversity of taxa (macroalgae, sessile and mobile invertebrate epifauna, infauna, fish, birds) that may be absent or at reduced densities in adjacent unvegetated soft-sediment habitats (Ruesink *et al.* 2005 and references therein).

Probably the main factors limiting the potential value of fouling habitat in the case of oyster farms would be the effect of enhanced sedimentation beneath rack structures, or sediment resuspension and physical disturbance from farming activities (Forrest & Creese 2006; see below). Accumulated shell, sticks and other inorganic debris from intertidal culture may persist for many years after the cessation of farming; the introduction of novel habitat created by such materials may result in long-term shifts in benthic community composition. There is likely to be site-specific variation in the significance of such effects according to environmental conditions, oyster species and density, and the extent of accumulation. Increasingly, regulatory authorities in other countries are stipulating management practices to mitigate such effects (e.g., requiring removal and land disposal of accumulated material).



**Figure 18.** Shell litter and sticks from abandoned oyster racks; a. Mahurangi Harbour (B Forrest); b. Waikare Inlet (B Howse, Northland Regional Council).

#### 3.3.3. Changes in seabed topography and sedimentation

Changes in seabed topography (in the order of a few tens of centimetres at maximum) have been described beneath oyster farms in several countries, including New Zealand (Ottmann & Sornin 1982; Everett *et al.* 1995; Forrest & Creese 2006). Such changes can be attributable to the accumulation of shell and inorganic debris, and erosion or accretion of sediment beneath and between farm structures (Forrest and Creese 2006). Sedimentation rates are elevated directly beneath cultures (Mariojouls & Sornin 1986; Sornin *et al.* 1987; Nugues *et al.* 1996) and in Mahurangi Harbour were almost three times greater than at control sites (Forrest & Creese 2006). However, Forrest & Creese (2006) suggested that effects on seabed topography were likely to be more related to changes in hydrodynamic conditions caused by the structures themselves rather than increased sedimentation rates. In New Zealand, sediment build-up to the top of Pacific oyster racks (Figure 19) can occur at sites where rack alignment is perpendicular to tidal currents and results in the entrapment of suspended sediments (Handley & Bergquist 1997). In such instances oyster leases have become un-useable and farming abandoned, with shell litter and debris still evident many years later (see Figure 18). The redistribution of sediments either into (Kirby 1994) or out of (Mallet *et al.* 2009) culture cites may also occur in relation to events such as storms that lead to large scale sediment mobilisation.



Figure 19. Sediment accumulation beneath oyster racks. (Photo: B Forrest).

## 3.3.4. Physical disturbance

At least two studies have implicated physical disturbance, in particular from vessel movements (*e.g.* propeller wash) and farm personnel walking between cultivation structures, as having a strong influence on benthic changes beneath oyster farm sites (De Grave *et al.* 1998; Forrest & Creese 2006). Forrest & Creese (2006) described an association between benthic macrofaunal composition and decreased sediment shear strength (increased ability for sediments to erode or resuspend) beneath Pacific oyster cultures in Mahurangi Harbour, which they suggested could reflect physical disturbance beneath racks (Figure 20). Physical disturbance is conceivably equally important as biodeposition and accumulation of shell material as a source of impact beneath cultivation areas, and perhaps more important where deposition effects are negligible. The relative importance of these two effects is yet to be rigorously evaluated, in part due to the difficulty of isolating influences of physical disturbance from the effects of biodeposition and subsequent organic enrichment of the seabed.





**Figure 20.** Oyster farm operations are a source of physical disturbance beneath oyster racks (Photo: B Forrest).

#### 3.3.5. Shading

Shading by farm structures could reduce the amount of light reaching the seafloor, with implications for the growth, productivity, survival and depth distribution of ecologically important primary producers such as benthic microalgae, macroalgae or seagrasses. Overseas studies have found effects on seagrass beneath oyster farms to be negligible (Crawford 2003), although at least one study has described adverse effects on seagrass beneath oyster racks and suggested shading as a possible cause (Everett *et al.* 1995). To our knowledge, the relative importance of shading versus other sources of seabed impact has never been conclusively established, and to do so would require targeted manipulative experiments. Despite the absence of clear evidence for adverse effects from shading, such impacts are nonetheless theoretically possible, as indicated by Hewitt *et al.* (2006) for a proposed oyster farm in Kaipara Harbour. Shading effects are conceivably of most importance where oyster farms are placed across seagrass and algal habitats in environments of relatively high water clarity, and in locations (*e.g.* well-flushed systems) where other ecological effects (especially those from sedimentation and biodeposition) are minimal. Shading effects on seagrasses and macroalgae can effectively be mitigated through appropriate farm placement.

## 3.3.6. Contaminant inputs

Operational oyster farms do not require the ongoing input of materials that could introduce trace contaminants to the marine environment, as can occur for example as a result of anti-fouling paints or synthetic feed inputs to sea-cage fish farms (Morrisey *et al.* 2000; Easton *et al.* 2002; Schendel *et al.* 2004). However, oyster racks may be constructed from treated timber (*e.g.* with copper, chromium and arsenic) that has the potential to leach contaminants into surrounding waters. Highly localised effects on sediments have been described in the vicinity

of marine pilings as a result of such leaching (Weis *et al.* 1993), consistent with expectations that trace metals that are released to the water column will rapidly bind to suspended sediment particles. Sediment binding of contaminants is likely to reduce the potential for toxic effects on associated biota (Förstner 1995), and the release of contaminants from treated timber in seawater is reported to decrease over time (Brooks 1996; Breslin & Adler-Ivanbrook 1998). Hence, this issue is probably of negligible significance in the case of oyster culture sites where wooden racks are used. We note that farmed shellfish are subjected to metals testing as part of water quality programmes, which would presumably detect biologically relevant accumulation should it occur. Nonetheless, there is an increasing trend overseas to use alternative construction materials, or to develop strict regulatory guidelines around the use of treated timber for oyster farm structures (*e.g.* DPI 2008).

## 3.3.7. Seabed effects following farm removal

Recovery rates of seabed communities from deposition-related enrichment effects of oyster farms are unknown, but are likely to be relatively rapid once farming ceases. Based on observations of temporal change in benthic effects from oyster farms in New Zealand (Forrest 1991, unpub.), and literature for mussel and fish farms (Mattsson & Lindén 1983; Karakassis *et al.* 1999; Brooks *et al.* 2003; Pereira *et al.* 2004), conceivable time scales of recovery range from a few months in well-flushed areas where effects are minor, to a few years in poorly flushed areas where moderate/strong enrichment has occurred. Accumulated shell, sticks and other inorganic debris from intertidal oyster culture may persist for many years after the cessation of farming (Forrest & Creese 2006); hence the introduction of these novel habitats may result in fundamental or long-term shifts in seabed community composition. There is likely to be site-specific variation in the significance of this change according to environmental conditions, oyster species and density, and the extent of accumulation. The wider ecosystem consequences of such habitat changes, and the ecological role of farm structures themselves, are discussed further below.

## 3.4. Oyster cultivation effects on the water column

#### 3.4.1. Effects of farm structures on currents and waves

Currents and waves play an important role in ecosystem function, particularly with regard to the transport of dissolved nutrients and seston (small particles and plankton) and nutrient exchange at the seabed-water interface. In relation to shellfish farming, currents and waves play an important role in the delivery of particulate matter and dissolved oxygen, and the flushing of wastes and associated nutrients into and out of the localised environment. If currents are not above a critical threshold to allow dispersion and resuspension of seabed sediments and associated detrital material from shellfish farms, for example, excessive accumulation of organic wastes and associated enrichment effects could occur.



Although there appears to be little published information for oyster farms, the farm structures and farm-related alterations to seabed topography (*e.g.* from shell accumulation) are likely to lead to effects on waves, currents and flushing characteristics in the vicinity of farm sites (Gouleau *et al.* 1982; Nugues *et al.* 1996; Hewitt *et al.* 2006). The structures themselves would be expected to baffle waves and currents, which in turn would enhance settlement and accumulation of particulate matter within close proximity of the farm. The effects of intertidal farm structures would be expected to have a proportionally greater effect on currents and waves than a subtidal structure, mainly due to the fact that the racks occupy a larger portion of the water column when submerged at high tide than a fully subtidal structure (see Section 2.4.1).

Literature for oyster reef habitats indicates that flow changes across the seabed may alter fluxes of materials (*e.g.* sediments) to adjacent habitats, and influence ecological processes such as patterns of dispersal and recruitment of invertebrates and fish (Breitburg *et al.* 2005; Ruesink *et al.* 2005). Effects of this general nature are also conceivable in the case of intertidal oyster culture, although specific differences can be expected given that the extent to which flows are modified will differ for different types of structure (*e.g.* because of differences in 'porosity' of structures as described in Section 2.4.1).

#### 3.4.2. Seston removal and alterations

Natural oyster reefs are considered to have the potential to improve estuarine water quality by filtering seston from the water column (Gottlieb & Schweighofer 1996; Ruesink *et al.* 2005; Grizzle *et al.* 2006). As a consequence, there is much interest in the restoration of degraded oyster reefs as a means of top-down control of phytoplankton densities in eutrophic estuaries (Newell 2004; Cerco & Noel 2007; Newell *et al.* 2007; Pomeroy *et al.* 2006, 2007). Whether intertidal oyster cultures have comparable benefits is unknown. On the basis that the filterfeeding capacity provided by oysters (and associated fouling) is likely to represent a considerable increase above and beyond that provided by filter-feeding benthos in the same area prior to cultivation, such effects are arguably possible.

The adverse effects of intertidal culture systems on water quality in estuarine environments are less well understood, but are likely to be relatively minor given that seabed enrichment is low and external contaminant inputs are minimal, as described above. We are unaware of any water quality data for New Zealand oyster farms that indicate adverse effects on water quality. The only cases of adverse water quality effects from oyster aquaculture arise from overseas examples of suspended cultivation where farms are over-stocked or located in poorly flushed environments. Early studies of suspended subtidal culture of Pacific oysters in Japan revealed adverse water column effects that were related to excessive biodeposition on the seabed (Ito & Imai 1955; Kusuki 1981). For example, Ito & Imai (1955) described seabed enrichment so severe that oyster culture areas became 'self-polluting' (*i.e.* leading to oyster mortality) as a result of dissolved oxygen depletion in the overlying water column and the associated release (from sediments) of hydrogen sulphide at toxic concentrations.



By contrast, a study in Marennes-Oléron Bay (a major Pacific oyster culture area in France) suggests that mortality occurs as a result of a range of factors, and not simply a negative feedback of water quality (Soletchnik *et al.* 2005). The findings of the latter study further indicate that the potential for adverse water quality-related effects in the case of intertidal culture is low, which is perhaps not surprising considering that intertidal farm sites are substantially or completely flushed approximately twice daily with every low tide. Any water quality effects associated with intertidal culture can undoubtedly be minimised by appropriate site selection and farm design (*e.g.* ensuring that farm structures are configured in a way that causes minimal retardation of flushing processes).

#### **Ecological carrying capacity**

Oysters can filter particles within the 4–100  $\mu$ m size range (Hawkins *et al.* 1998; Dupuy *et al.* 2000), hence can derive nutrition from phytoplankton (predominantly), detritus, bacteria, protozoa, zooplankton, and resuspended benthic microalgae (Le Gall *et al.* 1997; Dame & Prins 1998; Leguerrier *et al.* 2004). There has been considerable research into food depletion and modelling of ecological carrying capacity for oyster culture (Ball *et al.* 1997; Bacher *et al.* 1998; Ferreira *et al.* 1998) as well as for other bivalves and polyculture systems (Carver & Mallet 1990; Prins *et al.* 1998; Smaal *et al.* 1998; Gibbs *et al.* 2002; Nunes *et al.* 2003). Typically, this work has focused on phytoplankton depletion and maximum production capacity within growing regions. In this respect a number of indicators of carrying capacity have been used, in particular water residence time in relation to bivalve clearance and primary production time within a system (Dame & Prins 1998; Gibbs 2007). The literature in this field primarily addresses the role of natural or cultivated bivalve populations, whereas the filterfeeding activities of fouling organisms and other biota associated with shellfish cultures can also be functionally important (Mazouni *et al.* 2001; Mazouni 2004; Decottignies *et al.* 2007).

Influences from oyster aquaculture on estuarine carrying capacity are inextricably linked to the issues of nutrient cycling, seston depletion, and coupling between the water column and seabed. Interactions between shellfish cultivation, and the water column and seabed environments are complex, however, there is compelling evidence that bivalve aquaculture can affect nutrient cycling and the quantity and quality of seston across a range of spatial scales from local to system-wide (Prins et al. 1998; Cerco & Noel 2007; Coen et al. 2007). Control of Pacific oyster growth by phytoplankton availability has been described for subtidal floating culture systems in environments with long residence times such as Thau Lagoon in southern France (Souchu et al. 2001). In relation to elevated intertidal culture, Marennes-Oléron Bay has been described as "...one of the few systems where bivalve filter feeders have on two occasions been overstocked and overexploited" (Dame & Prins 1998). Marennes-Oléron Bay is a highly turbid system where bivalve clearance times are shorter than primary production and water residence times, and where resuspended benthic microalgae are an important food source (Dame & Prins 1998). There are anecdotal reports that Pacific oyster production in New Zealand estuaries has also been limited by carrying capacity, although this has not been definitively proven (Handley & Jeffs 2002, unpub.). The potential for such effects is invariably situation-specific and temporally variable. For example, the standing stock of phytoplankton and concentration of other SPM in estuaries is likely to be influenced by factors



operating from tidal time scales to longer term climatic events such as El Niño Southern Oscillation cycles (Dame & Prins 1998; Prins *et al.* 1998; Zeldis *et al.* 2000).

Evidence (albeit limited) that seston depletion from oyster culture can reach or exceed carrying capacity at bay-wide scales suggests that wider ecosystem effects are also possible. Such effects could conceivably arise not only as a function of depletion, but also through alteration in seston size spectra and plankton species composition. In turn this could affect the quantity and quality of food available to other consumers (Prins *et al.* 1998; Dupuy *et al.* 2000; Pietros & Rice 2003; Leguerrier *et al.* 2004), with consequences for local populations of higher trophic level organisms such as fish. Food-web modelling for Marennes-Oléron Bay predicted a shift from pelagic to benthic consumers as a result of intertidal trestle cultivation of oysters, reflecting SPM depletion in the water column and enrichment of benthic meiofauna (Leguerrier *et al.* 2004). It is conceivable, therefore, that intensive oyster cultivation could have flow-on effects throughout the food web; however, the scant literature in this field does not provide any evidence for adverse effects (see Sections 3.5.2 and 3.5.3).

#### 3.4.3. Seawater nutrient chemistry

The effects of intertidal oyster cultivation on seawater nutrient chemistry are poorly understood. Based on information from other bivalve culture systems, and natural or restored oyster reefs, it is evident that effects will be determined by processes involving filter-feeding and dissolved nutrient excretion, biodeposition and sediment remineralisation of nutrients, and loss of nutrients through oyster harvest (Newell 2004; Porter et al. 2004; Su et al. 2004; Prins et al. 1998). The production of dissolved (hence bioavailable) nutrients can occur directly via excretion by the oyster stock (Boucher et al. 1988), or indirectly via re-mineralisation and subsequent release from enriched sediments (Souchu et al. 2001). The subsequent effects of dissolved nutrient production on algal production involve complex interactions that are likely to be highly variable in relation to factors such as flushing, temperature, water clarity, stocking density, and the level of seabed enrichment. For example, although oysters may deplete phytoplankton, dissolved nutrients released from oyster excretion or sediment remineralisation have the potential to offset this effect by simultaneously stimulating phytoplankton production (Prins et al. 1998; Pietros & Rice 2003). Conversely, where filter-feeding by oyster reefs leads to locally increased water clarity (Cerco & Noel 2007), this may lead to increased production of benthic algae and seagrasses, thereby reducing the flux of dissolved nutrients to the water column and reducing phytoplankton production (Souchu et al. 2001; Newell 2004; Porter et al. 2004). For example, modelling by Cerco & Noel (2007) predicted that increased water clarity resulting from restoration of oyster reefs would lead to an increased biomass of submerged aquatic vegetation. For intertidal culture systems that are elevated, however, decreased sediment shear stress beneath racks, combined with turbulence induced by culture structures, may lead to enhanced sediment resuspension and high turbidity (Forrest & Creese 2006; Leguerrier et al. 2004). Clearly, nutrient cycling and related water quality attributes are influenced by complex environmental relationships and need to be further considered for intertidal culture in estuarine systems.



# 3.5. Wider ecological issues

#### 3.5.1. Habitat creation by farm structures

Marine farm structures and artificial structures in general, provide a three-dimensional reef habitat for colonisation by fouling organisms and associated biota (Costa-Pierce & Bridger 2002). In a manner similar to that described above for the accumulation of oysters and debris, such structures provide a novel habitat that can support a considerably greater biomass and density of organisms than adjacent natural soft-sediment habitats (Dealteris *et al.* 2004; *Crassostrea virginica* cages). It is also well recognised that assemblages on artificial structures can be quite different from those in adjacent rocky areas (Glasby 1999; Connell 2000), and comprise a diverse assemblage of macroalgae and filter-feeding invertebrates (Hughes *et al.* 2005). Hence, several studies have highlighted the role played by artificial structures within the ecosystem, such as increasing local biodiversity, enhancing coastal productivity, and compensating for habitat loss from human activities (Ambrose 1994; Costa-Pierce & Bridger 2002; Hughes *et al.* 2005). These types of ecological roles are recognised for natural oyster reef habitats and on-ground oyster culture, as noted earlier in this paper.

Recent evidence also suggests comparable roles for suspended subtidal oyster culture structures (Lin *et al.* 2007), intertidal trestles (Hilgerloh *et al.* 2001) or other intertidal structures used for oyster cultivation. For example, Dealteris *et al.* (2004) concludes that oyster cages used for the grow-out stage of *Crassostrea virginica* have a habitat value that is considerably greater than non-vegetated seabed and at least equal to seagrass. It is also evident that some intertidal culture systems provide a habitat that can be extensively colonised by naturalised oysters, as described for *C. gigas* in western France (Cognie *et al.* 2006).

## 3.5.2. Effects on fish

The aggregation of various fish species around marine farms and other artificial structures is well recognised (Relini *et al.* 2000; Gibbs 2004; Einbinder *et al.* 2006; Morrisey *et al.* 2006), reflecting the role of such structures offering shelter from predation, habitat complexity and a food source. In New Zealand, there has also been discussion of the potential negative effect of cultured oysters and mussels on fish populations, primarily due to the consumption of fish eggs (Gibbs 2004.). The association of fish with on-ground oyster culture (versus rack or stick culture as is the case in New Zealand) has been described in a number of studies (Grabowski 2004 and references therein), and in fact a wide suite of ecosystem services from the restoration of oyster reefs are recognised (Coen *et al.* 2007 and references therein). Similarly, in the case of on-ground clam culture in the United States of America, Powers *et al.* (2007) found that the emergent habitat provided by fouling of mesh bags led to densities of mobile invertebrates and juvenile fish that were elevated by comparison with adjacent sand flats, and comparable to seagrass beds.

Conceivably, therefore, the ecological role of elevated oyster farm structures, combined with habitat alterations from the deposition of oysters and associated debris, may affect fish

populations in a number of ways. However, a body of published information from primary literature comparable to that describing the effects of oyster reef or on-ground culture systems is unavailable for elevated culture systems, and the limited information available is equivocal. For example, Dealteris et al. (2004) describe a greater association with submerged aquaculture gear by some fish species but not others. Similarly, Dumbauld et al. (2009) cite Weschler (2004) who found no overall increase in fish richness or abundance adjacent to oyster racks, but a greater prevalence of structure-oriented species. Trophic modelling in Marennes-Oléron Bay represents one of few attempts to understand the wider ecosystem role of elevated intertidal ovster (Crassostrea gigas) culture (Leguerrier et al. 2004). These authors suggested that oyster cultivation could increase the food supply to fish, which was predicted to occur as a result of increased meiofaunal production. Similarly, increased turbidity (e.g., induced by erosion around oyster farm structures) may provide refugia from predation for small or juvenile life-stages of fish (e.g. Chesney et al. 2000; Leguerrier et al. 2004). A field mesocosm study of Pacific ovster cultivation effects in western France showed that the microhabitat created beneath trestles was more frequented by flatfish than adjacent homogenous habitat (Laffargue et al. 2006). More recently, an experimental scale deployment of oyster cages suggested that aquaculture gear could benefit populations of ecologically and economically important fish and epibenthic macrofauna in a way comparable to oyster reef habitat (Erbland and Ozbay 2008). Similarly, Lin et al. (2009) described an unexpectedly large decline in the biomasses of zooplanktivorous and piscivorous reef fish following the removal of an extensive area of high density oyster racks (up to 2932 racks/km<sup>2</sup>), although field-based sampling was limited in their study. These authors suggested that the oyster racks might have previously attracted reef fish by reducing predation or enhancing their food sources.

## 3.5.3. Effects on seabirds

#### Overview

There appear to be no New Zealand studies on the effects of oyster cultivation on seabirds. Based on overseas literature, and knowledge of mussel farm effects in New Zealand, it is evident that effects on seabirds conceivably arise due to the alteration of food sources, displacement of foraging habitat and as a result of disturbance (*e.g.* noise) related to farm activities (Kaiser *et al.* 1998, Connolly & Colwell 2005). The additional issue of entanglement has been widely discussed in New Zealand in relation to mussel farming (Section 2.5.3) and other forms of aquaculture or fishing practice (Taylor 2000a,b; Butler 2003; Bull 2007), but is unlikely to be an important consideration for intertidal oyster culture. Entanglement primarily arises where loose or discarded rope is present, hence is highly unlikely in oyster cultivation where fixed structures are primarily used. Similarly, the effects of plastic and other marine debris on seabirds have received attention both internationally and in New Zealand (Section 2.5.3) but we assume that such problems are minimal or non-existent in well maintained oyster farms.



#### Effects on food supply

The recognised role of marine structures in providing fish habitat (see above) could conceivably attract bird species to prey items. Griffen (1997, unpub.) suggested that the habitat enhancement provided by natural seabed oyster reefs may benefit some bird species (e.g. herons and other foraging birds) by providing an additional food supply. This view is supported by recent work in Argentina which examined the ecological role of naturalised Pacific oysters 20 years after their introduction (Escapa et al. 2004). The latter study revealed higher densities of local and migratory birds, and higher foraging rates, inside oyster beds compared with reference areas, which were attributed to greater prey availability. In the case of elevated intertidal culture, trophic modelling by Leguerrier et al. (2004) similarly suggested that birds could benefit from an enhanced food supply. Clearly, the consequences for higher trophic level animals that arise as a result of intertidal oyster farm effects on the nature, quantity or availability of their food supply will depend on consumer dietary preferences and their ability to adapt to changes induced by cultivation. Overall, the few overseas studies of oyster culture provide information consistent with other forms of aquaculture described overseas, suggesting an attraction of seabirds to culture areas for foraging fish and epibiota fouling structures, and even the cultured crop itself (Ross et al. 2001; Roycroft et al. 2004; Kirk et al. 2007).

#### Effects on foraging ability

Despite their potential to provide food sources for birds, the large areas of estuarine habitat that may be occupied by intertidal oyster farms means that they also have the potential to displace seabirds from foraging sites. The evidence for this is limited, and suggests effects will be species and situation-specific (see Dumbauld *et al.* 2009 and references therein). For example, Zydelis *et al.* (2006) suggested that shellfish culture racks or stacked bags/nets could block large intertidal regions from wading shorebirds such as oystercatchers, plovers, stilts and potentially dotterels. Certainly, for some bird species there is evidence from overseas studies of avoidance or a decreased association with oyster structures compared with open tidal flats (e.g. wintering shorebirds in California; Kelly 2001). Conceivably any bird species that avoid structured habitats may be susceptible to displacement effects. However, the published international studies directly investigating interactions between elevated oyster culture and birds provide little evidence for significant adverse effects.

A number of studies have found that instead of local bird species being excluded from foraging sites, their distribution was unaffected and they were actively exploiting cultured species as a food source (Carswell *et al.* 2006; Zydelis *et al.* 2006). A study of intertidal cultivation in California concluded that oyster long-lines did not negatively affect the foraging behaviour of most bird species, but rather enhanced it; there was a greater diversity of birds, and a greater density of some species of shorebird and wading bird, in long-line plots compared with controls (Connolly & Colwell 2005). In relation to trestle culture in Ireland, a preliminary study by Hilgerloh *et al.* (2001) found that oyster structures did not affect the feeding behaviour of birds. For most species, bird densities were lower in the farm area than a reference area; however, the authors recognised that this pattern may have reflected natural environmental differences. In addition to modifications to benthos, Hilgerloh *et al.* (2001) also



noted that macroalgae fouling the oyster trestles and associated small mobile gastropods provided a food source for some species.

#### Human disturbance

Disturbance of seabirds as a result of farm operations (*e.g.* noise and boat traffic) is an additional issue that should be considered for birds. Disturbance from noise and traffic does not appear to have been investigated in relation to elevated culture in estuaries, nonetheless New Zealand and overseas studies have reported that certain seabird species are more sensitive to human presence and disturbance (Goss-Custard & Verboven 1993; Butler 2003; Roycroft *et al.* 2004). Overseas, for example, Goss-Custard & Verboven (1993) found that oystercatchers were disturbed by the presence of humans in foraging areas, but were also surprisingly flexible in their ability to effectively redistribute their foraging activities. In New Zealand, Butler (2003) found that nesting king shags in the Marlborough Sounds were highly susceptible to disturbance by boats, leading to part or complete abandonment of nests and chicks. Varying levels of response to boat disturbance have been reported for several different shag species in New Zealand depending on location and their activity (Brown 2001; Lalas 2001). Lalas (2001) concluded, however, that boat traffic alone was not a significant disturbance factor to king shag foraging and/or resting activities. Consideration of effects from human disturbance on seabirds during the planning and site selection stages would assist in mitigating effects.

#### 3.5.4. Effects on marine mammals

#### Overview

There are a number of publications concerning interactions between marine mammals and aquaculture (Würsig & Gailey 2002; Kemper *et al.* 2003), from which it is apparent that potential effects on marine mammals include disruption of migration pathways (in the case of large cetaceans), displacement from feeding/breeding habitats, underwater noise disturbance, potential for entanglement and flow-on effects due to alterations in trophic pathways. For intertidal culture the significance of many of these potential interactions is unknown, and we are unaware of any New Zealand studies that have considered such issues for Pacific oyster cultivation. Conceivably, the potential for adverse interaction between intertidal oyster culture and marine mammals is minor in New Zealand, as there is probably minimal overlap between sites of intertidal cultivation and typical marine mammal habitat, as we discuss below.

#### Habitat exclusion

Watson-Capps & Mann (2005) reported significant habitat exclusion of Indian Ocean bottlenose dolphins (*Tursiops aduncus*) by pearl oyster farms in Western Australia, in a bay where racks were suspended or fixed to the seabed in relatively shallow water (~2-4 m). Tracks of individual dolphins showed that adult females tended to swim around or stay on the periphery of the farm boundary rather than travel through it. To our knowledge this is the only study that has focused specifically on oyster farms. Together with research on mussel aquaculture (Section 2.5.4), the study highlights that, while most cetaceans will not be completely displaced from a region as a whole, they may not utilise habitats occupied by aquaculture structures in the same manner as prior to a farm's establishment.

The nature of habitat exclusion will greatly depend on the type of culture method and the particular species of marine mammal present in the cultivation area. As such, site-specific knowledge is required in order to undertake a robust assessment of risks to marine mammals in a New Zealand context. We can only assume that the potential for overlap between marine mammals and intertidal oyster cultivation is minimal by comparison with subtidal suspended cultivation; farms located on tidal flats would only be accessible to marine mammals during periods of high tide. Nonetheless we recognised that marine mammals can use shallow habitats. For example, southern right whales (Eubalaena australis) and humpback whales (Megaptera novaeangliae) utilise New Zealand's shallow bays and protected beaches (e.g. they beach themselves to rub off fouling; D Clement, pers. obs.), and are vulnerable to obstacles within their migrating or feeding areas where they may remain for days to weeks (Kemper & Gibbs 2001; Kemper et al. 2003; Kraus & Rolland 2007). Field and captive studies have found that smaller dolphin species appear reluctant to swim through wooden structures or those with ropes (Kastelein et al. 1995; Watson-Capps & Mann 2005; Heinrich 2006). Pinnipeds (*i.e.* seals and sea lions) are perhaps the one marine mammal species that are unlikely to be excluded from habitats by the occupation of oyster farm structures. Effects of intertidal culture on mammals can be mitigated through farm placement and avoidance of areas heavily utilised by mammals.

#### Underwater noise

Underwater noise in the oceans has recently made headlines as a fairly widespread, yet largely unknown problem for marine mammals, particularly the larger whale species (Nowacek *et al.* 2007; Weilgart 2007; Wright 2008). Hence, knowledge of potential effects and noise-reduction technologies is likely to be important for present and future industry development. Currently, however, no New Zealand or overseas studies have specifically analysed noise production in association with aquaculture and marine mammals. In-water noise, especially vessel noise, is regarded as the primary issue of concern because of sound travel in the water column. Due to the intertidal location of cultivation, oyster farmers undertake most work during lower tidal conditions on racks when they are out of the water, in which case noise generation will not be a significant issue for marine mammals. The potential significance of in-water noise will depend primarily on the vessel traffic generated as a result of oyster farming relative to other activities. In general, we would expect that the level and persistence of any associated underwater noises with oyster farming (other than the initial construction) are likely to be insignificant relative to other forms of shellfish aquaculture and other underwater noise sources, such as commercial and recreational vessels.

## Entanglement

Oyster farming in New Zealand can occupy a significant area of intertidal habitat, as discussed above, effectively creating a novel obstacle that resident marine mammals have to actively manoeuvre around (Würsig & Gailey 2002; Markowitz *et al.* 2004; Watson-Capps & Mann 2005). Relative to some forms of aquaculture or fishing activities where rope or nets are used, the chances for marine mammal entanglement as a result of intertidal cultivation in New Zealand are probably minimal given that structures are either rigid wooden racks, or strong



plastic coasted wire under high tension (BST method, see Figure 16). Given the curious nature of most marine mammals, the entanglement risk associated with intertidal cultivation could conceivably include small dolphins or pinnipeds becoming wedged under rack structures. However, the likelihood of such adverse consequences is likely remote given that such effects have never been reported elsewhere in the world where intertidal cultivation is far more extensive.

#### **Indirect effects**

The potential for wider, more indirect ecosystem effects on marine mammals due to shellfish aquaculture include the interrelated issues of food-web interactions (Black 2001; Kaiser 2001; Würsig & Gailey 2002; Kemper *et al.* 2003), biotoxin and pathogen (disease) outbreaks (Geraci *et al.* 1999, Kaiser 2001), and antibiotic use (Buschmann *et al.* 1996; Kaiser 2001). While these potential indirect interactions between marine mammals and shellfish aquaculture have been considered in the literature (Würsig & Gailey 2002; Kemper *et al.* 2003), no indirect effects have been documented.

## 3.5.5. Biosecurity risks and biofouling pests

#### Background

The role of the oyster industry in the spread of non-indigenous species, biofouling pests, toxic or noxious microalgae (associated with biotoxin production and shellfish poisoning), and disease has long been recognised internationally. This is especially true in the case of macroscopic biofouling (Bourdouresque *et al.* 1985; Minchin 2007; Mineur *et al.* 2007; McKindsey *et al.* 2007), and associated organisms (*e.g.* Duggan 1979; Utting and Spencer 1992). A number of studies have also documented survival of toxic and nuisance microalgae as a result of aquaculture transfers (see mussel industry issues in Section 2.5.5), with overseas studies also highlighting the potential importance of oyster transfers (McKindsey *et al.* 2007). In fact, the introduction of *Crassostrea gigas* for aquaculture, and other oyster species to a lesser extent, is regarded as one of the most important pathways for the global spread of non-indigenous species (Verlaque 2001; McKindsey *et al.* 2007). Ruesink *et al.* (2005) estimated that more than 40% of non-indigenous marine species in Europe, the western United States, and North Sea may have been introduced by oyster aquaculture. Consequently, international transfers of shellfish for aquaculture are now subject to rigorous risk assessment procedures.

From a New Zealand perspective the sources of present day risk from oyster aquaculture are similar to that for mussel aquaculture, and relate to the potential for domestic spread of pest species by farming activities. Almost invariably, however, the initial introduction of the species to New Zealand will have occurred by vessel traffic (Cranfield *et al.* 1998). Below we discuss risks to natural ecosystems as a result of oyster cultivation and transfer, considering biofouling pests (and associated organisms) and diseases separately.

#### Spread of fouling pests via oyster aquaculture

The general ways in which aquaculture can contribute to the spread of pest organisms was outlined for mussel aquaculture in Section 2.5.5. Elevated or suspended structures (and

associated shellfish crops) provide ideal habitats for some species to proliferate at high densities (Carver *et al.* 2003; Lane & Willemsen 2004; Coutts & Forrest 2007), potentially acting as reservoirs for the subsequent spread of pest organisms. The association of pests with oyster cultivation is described for a number of algal species overseas such as *Codium fragile* spp. *tomentosoides*, *Sargassum muticum* and *Undaria pinnatifida* (Trowbridge 1999; Verlaque 2001; Mineur *et al.* 2007) and for biofouling invertebrates in New Zealand such as the sea squirts *Styela clava* and *Eudistoma elongatum* (Coutts & Forrest 2005, unpub.; Smith *et al.* 2007).

#### Existing and potential biosecurity risks from oyster farming in New Zealand

The actual role of oyster cultivation in New Zealand in the spread of pests is unknown, but some general comments can be made based on broader knowledge of aquaculture and marine biosecurity issues, including the discussion relating to mussel culture in Section 2.5.5. The spread of pest species from infested farm structures or oyster crops at local scales (e.g. within bays) is likely to be primarily driven by natural dispersal mechanisms; in particular the dispersal of planktonic propagules in water currents (Forrest et al. 2008). In contrast, spread at inter-regional scales often occurs via inadvertent transport with human activities (Minchin 2007). For example, infested material (equipment, seed-stock or crop) at an oyster farm, or associated service vessels, may be moved to other localities as part of routine aquaculture operations, as suggested for a proposed oyster farm development in Kaipara Harbour (Taylor et al. 2005). Based on studies with fouling pests associated with mussel culture in New Zealand (Forrest & Blakemore 2006; Forrest et al. 2007) and oyster culture overseas (Mineur et al. 2007), there is a high likelihood that associated fouling organisms will survive if such transfers occur without the application of treatments to reduce biosecurity risks. In this way, oyster farming activities may lead to the spread of pest organisms in natural habitats far from the point of first incursion, potentially leading to irreversible effects on natural ecosystems (Ruesink et al. 2005). We are unaware of any routine procedures in place for oyster farms in New Zealand to manage biofouling or other pest transfer risks. A heat treatment procedure for ovster spat was developed in response to the presence of the toxic phytoplankton species Gymnodinium catenatum in the Kaipara in 2000, but has not been used since and would apparently be invoked only if a similar incident recurred in a spat collecting area (Taylor et al. 2005).

#### Pacific oyster as a non-indigenous species

Pacific oysters cultured in New Zealand are non-indigenous, as is the case in many countries worldwide (Ruesink *et al.* 2005). After the arrival of Pacific oysters in New Zealand, they had spread to the Marlborough Sounds by 1977 (Jenkins & Meredyth-Young 1979) and to Tasman Bay by 1981, and thereafter continued to spread within these regions (Osborne 1991; Jenkins 1997). Their spread further south is considered to be limited by water temperatures that are too cool for successful reproduction. Naturalised populations of Pacific oysters throughout their New Zealand distribution can reach high densities in natural and artificial habitats of estuaries, ports and harbours. Similarly, Cognie *et al.* (2006) found that as much as 70% of the oyster stock in a Pacific oyster growing area of the French Atlantic coast comprised naturalised rather than cultured oysters.

While Pacific oysters may be invasive primarily in rocky habitats and artificial structures, there is also evidence that they can invade soft-sediment estuarine habitats both overseas (Cognie *et al.* 2006; Smaal *et al.* 2009) and within their distributional range in New Zealand (Jenkins 1997; Forrest B, pers. obs.). The dense aggregates of Pacific oysters which form through high spat settlement on intertidal shores are a familiar site in some areas. Pacific oyster reefs in New Zealand can accumulate mud, and sharp oyster shell can degrade coastal recreation (Hayward 1997). Naturalised Pacific oyster populations may also displace native species in New Zealand (Dromgoole & Foster 1983). Based on the many overseas studies cited above highlighting the structural and functional role of oyster reefs or cultures in natural ecosystems, it can be expected that dense aggregations of naturalised oysters have the potential to lead to significant ecological changes (arguably both adverse and beneficial) in habitats where they establish (*e.g.* as described for Pacific oysters in Argentina by Escapa *et al.* 2004).

## 3.5.6. Disease

There have been no documented (OIE listed, OIE 2001) serious parasites/pathogens of Pacific oysters in New Zealand (Diggles *et al.* 2002). Nevertheless several diseases and parasites associated with New Zealand Pacific oysters have been reported, most of which are also globally ubiquitous and pose some commercial threat to oyster production (especially in hatcheries). These include various species of flatworm and mud-worm (Handley & Bergquist 1997; Handley 2002) and herpes virus, which infects oyster larvae and spat. Summer mortalities of oyster seed have been linked to herpes virus in California but a causal association has not been confirmed (Friedman *et al.* 2005). More detailed information on these and other diseases documented overseas is provided in Appendix 2.

To date, New Zealand farmed Pacific oysters have not suffered significant or unexpected effects from indigenous pathogens such as APX, *Bonamia exitiosa*, rickettsia and digestive epithelial virosis. This supports findings reported elsewhere that Pacific oysters appear more resilient to some diseases (Elston 1993) suffered by other oysters. In the light of this and since there have been extensive pathology surveys on New Zealand Pacific oysters - with negative results - it is inferred that culture of pre-existing *C. gigas* in New Zealand is unlikely to pose a pathological threat. However, any new importation of *C. gigas* stock should be subject to examination and be sourced from a documented disease-free area. This is suggested because observations from overseas indicate that there is a risk of spreading disease via introduction of oysters for culture – particularly from Pacific oysters. See Appendix 3 for examples.

Although New Zealand may lack some of the diseases identified overseas, congenerics and others of close affiliations do occur in New Zealand waters and could be similarly affected. It follows that should New Zealand *C. gigas* suffer an incursion by an exotic disease, it is possible that oyster farms could assist in the spread of disease to other molluscan species. Despite this possibility, the effect of non-native species can be unpredictable. For example, Thieltges *et al.* (2008) reported that the presence of introduced Pacific oysters (*C. gigas*) and American slipper limpets (*Crepidula fornicata*) mitigated the effects of a trematode parasite on

blue mussels (*Mytilus edulis*). It appears that the introduced oysters diverted the trematodes from their usual native hosts, thus reducing infection levels.

## 3.5.7. Genetic interactions with naturalised oyster populations

Aquaculture of oysters, and hatchery production of spat, invariably raises the issue of genetic interactions with naturalised populations. The pertinent issues and concerns in this regard, were discussed for mussels in Section 2.5.7 and in detail in Appendix 3. In the case of Pacific oyster cultivation, ecological effects on naturalised populations are not as relevant since Pacific oysters are non-indigenous to New Zealand. Furthermore, recent advances in breeding and the future production of triploid oyster spat that are sterile will likely eliminate effects associated with genetic interactions between naturalised, farmed and hatchery populations.



# 4. DEVELOPING AND POTENTIAL NON-FINFISH SPECIES

## 4.1. Overview

Subtidal cultivation of Greenshell<sup>TM</sup> mussels and intertidal cultivation of Pacific oysters currently dominate the non-finfish aquaculture industry in New Zealand. Although a number of other non-finfish organisms have recognised aquaculture potential (Table 4). At the time of writing, sea-based 'commercial' farming of other species was limited to two small paua (abalone) farms, and small scale operations for seaweeds and oysters (excluding intertidal), most of which are being co-cultured with mussels. A few other species (*e.g.* crayfish, kina, paua) are sometimes held in sea-based cages after being harvested from the wild, to either synchronise the sale, or 'condition' the animals, to achieve optimum market value. Although not technically aquaculture, most of the issues discussed for Greenshell<sup>TM</sup> mussel and Pacific oyster culture are equally applicable to these grow-out or 'sea-ranching' operations.

Experimental research is being conducted with a broader variety of species, some of which are undergoing growth trials on existing farms, usually alongside established mussel cultures. In addition to those noted above, other species include scallops, blue mussels, sponges, sea cucumber and geoduck (Table 4). In many instances, the commercial sensitivity of new species development is such that information is not freely disseminated. Moreover, the size, scarcity and relative newness of these industries generally means that any associated environmental effects have not been described or are not yet fully expressed; hence related literature is sparse or non-existent.

It is assumed that many of the environmental effects that arise from cultivation of these other non-finfish species will be common among farming that involves similar cultivation methods (e.g. backbone suspended culture) and/or involves organisms with similar feeding strategies (e.g. filter-feeding bivalves). This is because most of the effects described in Sections 2 and 3 stem from either feeding and waste products or the physical presence of the structures themselves. In the absence of information on the ecological effects of potential culture species, we provide some general guidance on the likely nature and magnitude of effects based on information from comparable species or culture methods. To facilitate this assessment, we broadly group the candidate species according to their trophic level (*i.e.* position in the food chain) as this has implications for the nature of farm wastes that are generated. This approach allows us to consider potential effects on the seabed and in the water column and how they may differ among species (e.g. greatest seabed effects are likely to occur for species that require external food inputs such as artificial diets). Only general comments can be made with regard to broader ecological issues, because the culture methods and environments are yet to be defined, and/or the issues are highly species-specific and poorly understood (e.g. disease issues, genetic interactions between cultured and wild stocks).

Classification	Species	Current status	
	Pacific oyster (subtidal) (Crassostrea gigas)	Experimental trials being conducted on existing mussel farms. (image source: www.fish.gov.au)	
Filter feeders	Flat oyster (Ostrea chilensis)	Experimental. Recent Government investment to research hatchery and grow-out methods. (image source: www.fish.govt.nz)	
	Scallops (Pecten novaezelandiae)	Tried on small scale commercial. None successful to date. Subject to experimental trials.	
	Blue mussel (Mytilus galloprovincialis)	Usually considered a pest, but small volumes harvested as an incidental by-product of Greenshell <sup>TM</sup> mussel culture.	Distanti
	Sponges (Bath sponge <i>Heterofibria</i> )	Experimental trials being conducted in Marlborough Sounds	
Grazers	Paua ( <i>Haliotis iris</i> )	At least two small scale commercial farms growing/holding paua in barrels (image source: www.fish.gov.au)	
	Kina (Evechinus chloroticus)	Experimental trials being conducted by NIWA (PGST Contract C01X0301) (image source: www.seafood.co.nz)	
Deposit feeder	Sea cucumber ( <i>Stichopus mollis</i> )	Not farmed commercially. Experimental trials. Potential as integrated culture species.	
Scavenger	Crayfish ( <i>Jasus edwardsii</i> )	Research and experimental trials being conducted by NIWA (PGST Contract C01X0301) (image source: www.lobster.co.nz)	
Primary producer (seaweed)	Macrocystis pyrifera, Undaria pinnatifida	Small level of farming/wild harvest of <i>Macrocystis</i> . For <i>Undaria</i> Industry development limited 'unwanted species' designation. (image source: www.starfish.govt.nz)	Underia

# **Table 4.**Minor and experimental culture species in New Zealand, with reference to the current status of the<br/>activity in New Zealand.



# 4.2. Commonality of effects among species

The reviews of Greenshell<sup>TM</sup> mussel and Pacific oyster effects in this report, and other reviews of the effects of finfish and other types of aquaculture in New Zealand (Cole 2002; Forrest *et al.* 2007) highlight that for different species, culture methods and culture environments, a similar suite of ecological issues arises. In the following section we describe in broad terms for each of these main issues, how the nature and magnitude of ecological effects may differ among the candidate species, different culturing methods and new environments.

In overview, those issues that we consider to be primarily species-specific are seabed and water column effects, disease and genetic interactions. Issues dictated more by culture method or farm management are likely to be effects on higher trophic level organisms (*e.g.* seabirds, marine mammals), production of contaminants, and the creation of novel habitat and spread of associated pests. Below we discuss examples that illustrate these general points, and make comparisons with the known effects of shellfish (Sections 2 and 3) and finfish (Forrest *et al.* 2007) aquaculture in New Zealand (as appropriate) to provide some feel for the nature and magnitude of effects that could arise with new species. In Section 4.3 we provide additional detail on actual or potential effects for each of the candidate species.

#### Seabed and water column effects

The propensity for benthic and water column effects can be roughly determined by the diet and feeding mechanism of the candidate species, their waste production and their likely culture method. The cultivation of organisms that require external feed inputs (e.g. crayfish, paua) are likely to produce more waste products than cultivation of species that do not rely on external feeds (see Table 4). The combination of excreted waste and uneaten feed has a relatively high potential to adversely affect the local seabed, as is evident in the case of salmon farming (Forrest et al. 2007). Dissolved waste products (e.g. ammonia and nitrate) may also stimulate algal production in the water column. By contrast, the cultivation of organisms (e.g. bivalves and sponges) that filter food (e.g. phytoplankton) from the surrounding water column and deposit organic waste on the seabed has less potential to cause adverse effects. Nonetheless, in high density culture situations filter-feeding pressure can be sufficient to cause localised depletion of phytoplankton, and overseas studies reveal the potential for relatively pronounced seabed effects in certain environments. How this potential varies among filter feeding species is discussed in more detail in Section 4.3.1. The cultivation of seaweeds (macroalgae), which function at a lower trophic level and utilise only dissolved nutrients and sunlight presumably leads to minimal ecological effects.

## Contamination

The two main sources of contamination from sea-based aquaculture stem from additives in feed inputs (if required), and leaching of chemicals from farm structures or structure coatings, such as antifouling paints or treated timber. Documented cases of contamination arising from forms of aquaculture other than fish farming (see Forrest *et al.* 2007) have been negligible. Contamination is considered a culture method specific issue due to its likely origin in farming structures and feeds. However, the types of structures used would need to be vastly different from those presently used in the mussel or oyster farming industries for any issues to arise.



Most of the other species considered here are based around modified long-line type methods (and materials) and as such are unlikely to induce appreciable chemical contamination. However, consideration is given to the likely feed input for farm species that require an externally derived food input (*e.g.* crayfish, paua and kina).

#### Effects on fish, seabirds and marine mammals

The effects of farming the species listed in Table 3 on wild fish populations is poorly documented; however, is likely to be less pronounced than those associated with finfish aquaculture, which involves addition of an external food source to the environment (e.g. Dempster *et al.* 2002, 2004). The effects of large offshore sites (*e.g.* >1000 ha) warrant separate consideration given the scale of these developments compared with existing operations. Furthermore, offshore developments tend to be situated within range of various inshore commercial fish species (see Sections 2.5.2 and 6.1.1).

Effects on seabirds and marine mammals are likely to depend primarily on culture method, farming practice and environment (*e.g.* extent of overlap with critical habitat), and cannot be predicted in the absence of specific information. In the case of marine mammal entanglement, the review in Sections 2.5 and 3.5 revealed that the risk of entanglement is related to culture method and farm management. For example, the use of fixed structures or lines under tension is less likely to lead to entanglement than loose rope or line. At this stage, the specific nature of the culture methods for most candidate species, and the environments in which they will be cultivated are unknown.

#### Biosecurity risks and biofouling pests

The creation of novel habitat for fouling organisms and associated biota is well recognised for marine farms and other artificial structures in the sea, as discussed in preceding sections of this report. To some extent the nature of the 'reef' community associated with such structures will be related to culture method, since the size of the structure, construction materials, and orientation of structures, are likely to facilitate colonisation by different types of assemblages (Glasby 1999; Connell 2000; Glasby & Connell 2001). Moreover, the association of non-indigenous or pest organisms with marine farms and artificial structures is well recognised (Glasby 1999). Marine pest 'risks' arise mainly transfer of seed-stock or equipment that can move pests beyond natural barriers to their dispersal. Hence, biosecurity is also a species-specific issue that will be exacerbated by industries that 1) have a high degree of transfer between regions, 2) involve species not naturally widespread or indigenous to the bay/region, and 3) do not have biosecurity protocols for stock or equipment transfer and management in place.

#### Disease

High density cultivation of organisms raises the potential risk of disease transmittance to the surrounding environment. The risks associated with farming of these minor and potential species listed in Table 4 will be similar to those outlined for mussels and oysters (see Sections 2.5.6 and 3.5.6), although the diseases themselves and their pathology will often be unique to a



given species. Detailed information on diseases that affect the health of cultivated non finfish species is provided in Appendix 2.

#### Genetic interactions between cultured and wild stocks

Genetic issues associated with sea-based aquaculture arise from either the transfer of wild caught stock between regions or the transfer of hatchery-reared stock to the wild, both of which have the potential to irreversibly alter genetic profiles of wild populations. Risk of this occurring is dependant on some species-specific factors such as: the level of genetic structuring within the species (low level of structuring corresponds to low susceptibility), local genetic diversity (wild spawning mechanisms ensure adequate genetic mixing), and, in the case of established culture species (*e.g.* mussels), the pre-existing levels of transfers. Factors specific to risks associated with the transfer of hatchery reared stock to wild populations mostly concern the potential for creating a bottleneck in the gene pool. Risks are greatest if the introduced stock is genetically narrow, there is a high level of structuring within the wild population, and the farmed stock is introduced in large numbers (numerical pressure).

# 4.3. Specific effects associated with candidate species

## 4.3.1. Filter-feeding bivalves

Cultivation of filter-feeding bivalves, other than Greenshell<sup>TM</sup> mussels and Pacific oysters, is currently limited to experimental trials, research and incidental by-catch (*e.g.* blue mussel). Species with feasible techniques and/or the most potential include oysters (flat oysters (*Ostrea* (formerly *Tiostrea chilensis*) and subtidal cultivation of Pacific oysters (*C. gigas*), scallops (*Pecten novaezelandiae*) and the blue mussel (*M. galloprovincialis*). Note that Section 3 focused on intertidal cultivation of Pacific oysters and that subtidal cultivation is still evolving and hence is considered here.

There are a number of other bivalves with potential, such as cockles (*Austrovenus stutchburyi*), geoduck (*Panopea zelandica* and *Panopea smithae*), toheroa (*Paphies ventricosa*), tuatua (*Paphies subtriangulata*) and several surf clam species (*Mactra* spp., *Dosinia* spp. and *Bassina* sp.). However, these species occur naturally within substrate or a 'sediment matrix', which is difficult to artificially reproduce off the bottom and adversely affects the economic feasibility of culturing these species. With the exception of the geoduck, which has exceptionally high economic value, the culture of sediment-dwelling species is unlikely to extend beyond enhancement of wild populations in the near future; and as such, they are not considered any further in this review.

## Transferability of effects among bivalves

The commonalities of the physiology and likely culture techniques of the bivalve species considered here are such that many of the issues described for the Greenshell<sup>TM</sup> mussel or intertidal cultivation of Pacific oyster are relevant. Filter-feeding bivalves all obtain their nutritional requirements by filtering out suspended organic particulates (primarily phytoplankton and detritus) from the water column. Some species such as scallops are more



dependant on benthic microalgae that become resuspended near the sediment surface (Gillespie *et al.* 2000; Keeley 2001; Gillespie 2008); however, they would rely on similar food sources as mussels when cultured within the water column. Bivalves process particulates from the water column and release both 'faeces' and undigested material called 'pseudofaeces', which are slightly heavier than water and sink to the seabed. The potential for localised phytoplankton depletion would be dependent on the clearance rate for a given species and the densities at which they are farmed. The potential to cause organic enrichment of habitats on the seabed would in turn be influenced by the rate at which particulate matter was processed and deposited on the bottom.

All of the bivalves considered here are also broadcast spawners, which means they release gametes directly into the water column, where dispersal range is determined by duration of gamete viability and current speeds. Accordingly, there is potential for progeny to colonise adjacent natural habitats and for mixing of farmed and wild populations. The scope for this issue arising is both species-specific and site/situation-specific and will need to be considered as part of any assessment of environmental effects. Of particular importance is the means of procuring stock or 'seed', where wild sources inherently maintain genetic diversity. Significant outbreeding depression has not been observed with current primary (*i.e.* mussels) and secondary species (*i.e.* scallops, oysters) and the same is likely to be true for analogous forms of bivalve aquaculture.

#### Relative potential effects of main bivalve species

The most likely culture methods for oysters (Pacific and flat), blue mussels and scallops, employ variations to the suspended culture techniques that have been developed around the mussel industry. The relative environmental effects of culturing different densities of these bivalve species in suspension were considered recently by Gibbs *et al.* (2006) using available, pertinent physiology literature (Table 5). A hazard assessment was used to identify the major environmental interactions between bivalves and the surrounding marine environment and this highlighted several major risk pathways, several of which were through the feeding and excretory behaviour of the bivalve crop. Marginal differences between the transfer of material by the different species were investigated using a range of feeding models and environmental data from the Marlborough Sounds and Glenhaven Aquaculture Centre (Nelson). The key result was that mussels generally appear to exhibit the highest clearance and excretion rates of the bivalves considered (Figure 21). Similarly, biodeposition intensity greater than 400 g/day/1000 individuals occurred most frequently in mussels (40%) followed by, scallops (33%), cupped oysters (29%), flat oysters (11%), and finally clams/cockles (6%).



Table 5.	Summary of literature used in Gibbs et al. (2006) comparison of density dependant effects
	associated with culturing potential bivalve culture species.

<b>Group/species</b>	References
Clams/cockles	Pfieffer <i>et al.</i> (1999), Teiaoro 1999, Bacon <i>et al.</i> (1998)
Flat oysters	Haure et al. (1998), Rodhouse & O'Kelly (1981)
Cupped oysters	Ren <i>et al.</i> (2000), Bacher & Baud (1992), Barillé <i>et al.</i> (1997), Bougrier <i>et al.</i> (1995)
Scallops	Keeley (2001), Laing (2004), Teiaoro 1999
Mussels	James et al. (2001), Hawkins et al. (1999), Cawthron unpublished

Overall, the model indicated that the substitution of Greenshell<sup>TM</sup> mussels with any of the other alternate species/groups proposed was unlikely to increase either the clearance of the surrounding water, the biodeposition of suspended matter or the amount of dissolved ammonia through excretion. In fact, on an equivalent numbers basis substitution with any of the alternate groups may reduce these interactions, especially where either flat oysters or clams/cockles are considered. The study concluded that other bivalves species such as scallops, oysters and cockles may be cultured at stocking densities equivalent to those used for mussels without posing additional risk to the marine environment.

The hazard assessment also identified that farming structures can potentially lead to changes in the surrounding environment through the alteration of water flows. Scaling analyses were performed that highlight the relative differences in cross-sectional areas posed by different farming methods (Figure 22). The results from this analysis indicate that the present mussel farming practices occupy a greater cross-sectional area by comparison with the other methods that are presently used. Hence there is little evidence to suggest that the stocking densities of other bivalve species using different growing techniques should be more overly restricted by comparison with present marine farming practices.





**Figure 21.** Comparative histograms showing the range and relative frequency of predicted clearance rates by bivalve group for the Glenhaven Aquaculture Centre ponds (from Gibbs *et al.* 2006).





**Figure 22.** Scale drawings of potential culture configurations for the other bivalve species, considered by Gibbs *et al.* (2006). Specifications based on industry advice in 2006.



#### Pacific oyster (Crassostrea gigas) – subtidal culture

In New Zealand, Pacific oysters are traditionally cultured in the intertidal zone on racks or in baskets (see Section 3). Recently, there has been considerable interest in culturing Pacific oysters subtidally using a variety of innovative structures suspended from conventional long-lines. While still considered experimental, it is generally accepted that subtidal cultivation of Pacific oysters will be further developed.

Environmental effects arising from subtidal oyster cultivation are likely to be analogous to those described for subtidal mussels (see Sections 2 and 3) with perhaps some differences according to species-specific predisposition to diseases, genetic conditioning, biosecurity issues and ability to induce water column and benthic effects. As discussed in the previous section, the propensity for Pacific oysters to induce benthic or water column effects is expected to be comparable or less than that of other bivalve species such as Greenshell<sup>TM</sup> mussels. Disease and genetic issues for the Pacific oyster have been discussed in Section 3 and Appendix 2 and most of these findings are directly transferable to subtidal culture. The potential for disease transfer between oysters and mussels would need to be considered in cases of co-culture. It is also worth recognising that Pacific oysters are non-indigenous to New Zealand and considered by some to be an invasive pest species, mostly due to the access hazard they create around rocky shorelines. An important consideration with the development of this industry should be to ensure that it does not facilitate the spread of Pacific oysters to areas or regions that it is yet to colonise.

#### Flat oyster (Ostrea chilensis)

Although not commercially cultivated in New Zealand, *O. chilensis* has strong potential due to its highly regarded edibility that commands premium prices. Development of this industry has been hampered by difficulties in the larval production stages of culture (Jeffs & Creese 1996). Growth trials are presently being conducted using a modified long-line method in the Marlborough Sounds and the Cawthron Institute recently received Government funding to develop hatchery seed-production and grow-out techniques. Small commercial volumes are being produced on two farms in Southern New Zealand (Pers. Comm. M. Mandeno). If successful, flat oysters may become an important aquaculture species in the near future. Culture methods will most likely employ one of a variety of existing basket, purse or tray systems (see Figure 22) suspended from long-lines within existing AMAs.

Like *C. gigas*, *O. chilensis* belongs to the super-family Ostreacidae, and as such, shares similar physiological characteristics and presumably, a similar propensity for environmental effects. Although commercial culture techniques for the species are yet to be formally established, they are expected to be similar to those used for *C. gigas* and therefore create similar issues with respect to creation of novel habitat and associated wider ecological issues. Disease and genetic issues are however, likely to be species-specific and warrant some consideration. Unlike *C. gigas*, *O. chillensis* is native to New Zealand, which has potential implications for the transfer of disease and genetic material from farmed stock to wild populations. Wild (commercially fished) populations of *O. chilensis* have also experienced disease problems which may arise in a cultured environment. A summary of the disease literature is provided in Appendix 2.



#### Scallop (Pecten novaezelandiae)

The New Zealand scallop *P. novaezelandiae* belongs to the Pectenid super-family and like mussels and oysters, obtains its energetic requirements by filter-feeding (phytoplankton, diatoms and organic detritus). In the wild, scallops tend to be less aggregated and more widely dispersed than mussels and exist partially immersed in soft sediments. Over the last 20-30 years repeated attempts have been made to culture the species in suspension, but this habitat requirement has been difficult to replicate or overcome (Hayden 1998; Keeley 2001) and as such, commercially feasible culture methods have yet to be established. *P. novaezelandiae* does, however, grow off the seabed (Keeley 2001; Heasman *et al.*, in prep.) and is the subject of ongoing culture trials, and it is conceivable that the species will be cultured in the near future.

Environmental effects arising from scallop culture are likely to be analogous to those described for mussels. This is particularly true for most of the wider ecological issues. An exception to this may be the scope for shell drop-off, as scallops are mostly likely to be culture in cages or attached to substrates, in which case drop off would be minimal compared with mussels. Issues considered specific to this species include disease and ability to induce water column and benthic effects. The propensity of *P. novaezelandiae* to induce benthic or water column effects is compared with that of *P. canaliculus*, oysters and cockles in Section 4.2. Otherwise, actual studies that consider direct environmental effects of scallop culture are rare. An exception to this is Zhou *et al.* (2006), which described significant filtration pressure and enhanced biodeposition associated with intensive culture of the overseas scallop *Chlamys farreri*. Interestingly, the same study concluded that intensive scallop culture could be advantageous ecologically, by functioning as a biofilter and potentially mitigating eutrophication pressures. However, this argument only holds if over-enrichment is an issue, such as in Sushili Bay, China, where that study was conducted. By comparison, potential sites in New Zealand are nutrient poor and the culture methods far less intensive.

In terms of disease, experience from wild populations indicates that while numerous parasites are found in New Zealand scallops, none appear to present a serious threat and only a few have pathological significance. Among the more significant are digestive epithelial virosis (DEV), Rickettsia-like organisms (RLOs) and a new unidentified inclusion (Webb & Duncan 2008). No OIE (2000) listed diseases were reported. Disease prevalence and pathological significance in scallops are discussed in more detail in Appendix 2.

#### Blue mussel (Mytilus galloprovincialis)

Blue mussels (*M. galloprovincialis*) are cultured incidentally in significant quantities with Greenshell<sup>TM</sup> mussels on farms in the Marlborough Sounds. Although blue mussels are cultivated overseas, they are considered a nuisance species in New Zealand because of the issues they create (competition for resources, on-site fouling management, post-harvest sorting and disposal) for the culture of Greenshell<sup>TM</sup> mussels on which our marketing brand is based. However, a small volume of blue mussels do get processed and sold on the local market and if desired, *M. galloprovincialis* could easily become a major culture species. Its obvious

suitability to conventional Greenshell<sup>TM</sup> mussel long-line culture techniques means it would inevitably be farmed in a very similar manner.

Blue and Greenshell<sup>TM</sup> mussels are both filter-feeding bivalves and share the same family, Mytilidae, and have an appropriately similar physiological make-up. Blue mussels tend to have smaller maximum and harvest sizes, and are reported to have similar food demands, as determined by clearance rates, rejection rates and energy requirements (Navarro *et al.* 1996; Hawkins *et al.* 1999). The absence of any monocultures of *M. galloprovincialis* in New Zealand means that literature pertaining to species-specific environmental effects has not been produced. However, given that the species is often (inadvertently) co-cultured in significant quantities on Greenshell<sup>TM</sup> mussel farms, any observed environmental effects associated with those farms must at least be partially attributable to *M. galloprovincialis*. Moreover, overseas studies of environmental effects resulting from suspended culture of *Mytilus* spp. (Hatcher *et al.* 1994; Heasman *et al.* 1998; Stenton-Dozey *et al.* 2001; Chamberlain *et al.* 2001) identify comparable benthic and water column issues to those described in Section 2; although in some instances they are more acute due to the relatively intensive raft culture techniques.

There is a general paucity of information surrounding disease threats associated with M. *galloprovincialis* in New Zealand. What information does exist, indicates little in the way of pathological problems, with the exception that invading *Mytilus* could act as a vector for facilitating the establishment of other serious exotic diseases. The species also has the potential to harbour viruses that may affect other species such as fish (Kitamura *et al.* 2007), but this is yet to be observed in New Zealand. Although poorly described, parasite fauna of *M. galloprovincialis* are likely to be comparable to that of the Greenshell<sup>TM</sup> mussel. More information relating to the pathology of the blue mussel is provided in Appendix 2.

## 4.3.2. Sponges and other low trophic level filter feeders

This group encompasses the 'simple' or low trophic level species, which filter particulate matter (bacteria and phytoplankton) from the water column. These organisms are typically sessile (fixed in place) and often encrusting by nature. Potential taxa within this group include a variety of sponges, ascidians, and hydroids, some of which have been examined for pharmaceutical properties (Page 2003), or physical properties in the case of the bath-sponge *Spongia (Heterofibra) manipulatus.* Sponges have also been considered for use in integrated culture systems as bioremediators of pathenogenic bacteria (Fu *et al.* 2005). New Zealand's experience with culturing these minor species is extremely limited and information pertaining to their culture is accordingly sparse. One exception is some recent work done with the bath-sponge in the Marlborough Sounds (Handley *et al.* 2003; Kelly *et al.* 2004), but no literature exits pertaining to the environmental effects of farming this or any other similar species. Hence the environmental effects of commercial culture of these types of organisms are poorly understood.

By comparison to other filter-feeders (*i.e.* bivalves), sponges exist naturally in low densities, but individuals can be large, and have high individual clearance and filtration rates. For

example, a sponge with an 8 cm osculum (opening) can process 180 L seawater hr<sup>-1</sup> and 1300 mg suspended solids hr<sup>-1</sup> (Yahel *et al.* 2007) compared with 1-10 L/hr<sup>-1</sup> and 1-1000 mg suspended solids hr<sup>-1</sup> for an individual adult Greenshell<sup>TM</sup> mussel (Hawkins *et al.* 1999; James *et al.* 2001). In natural densities, sponges have been observed to alter the composition of the suspended particulates in the water column up to 0.75 m above the seabed (Yahel *et al.* 2007) so water column carrying capacity issues may be pertinent if cultured intensively. Presumably, sponges produce waste products, however, no information could be found detailing rates and composition of biodeposits. Commercial culture methods are yet to be established, but trials with *S.* (*H.*) *manipulatus* in the Marlborough Sounds utilised a modified lantern net design (Kelly *et al.* 2004). Many sponges favour exposed, or high flow environments that tend to coincide with rocky coastlines and reef habitats; hence, there is potential for overlap of aquaculture requirements with high value ecological habitats. There is also potential for biosecurity issues given the invasive nature of some ascidians, but these are negligible if the culture organism is native to the site.

## 4.3.3. Grazers and deposit feeders

This group of organisms is united by the fact that they either graze benthic algae (micro and macro) and/or eat detrital matter accumulated on the seabed.

## Paua (Haliotis iris)

Paua (H. iris) aquaculture in New Zealand is mostly conducted in land-based systems, which can accommodate all phases of production (spawning, larval rearing, seed production and grow-out). Many hatcheries also now produce juveniles for reseeding and replenishment of wild stocks (Keeley et al. 2006). Both of these activities are beyond the scope of this report; however, there are presently at least two marine-based aquaculture operations (one in the Marlborough Sounds and one on Banks Peninsula) growing small amounts of paua, either for pearl cultivation or grow-out for harvest. Sea-based containment systems in New Zealand typically comprise barrels suspended from conventional backbone lines. Paua are grown inside the barrels where they are fed brown and/or red macro-algae (e.g. Macrocystis pyrifera, Lessonia sp., Durvilliae and Pterocladia spp.) and in some instances specially designed feed pellets. On at least one of the two existing farms, consent conditions specify that they can only feed paua the macro-algae that naturally colonises the farming structures, *i.e.* no artificial feeds or introduced biomass. Indeed, there is recognised potential for growing paua in a balanced, integrated co-culture system with algae, whereby minimal waste products are produced (Langdon et al. 2004). Abalone, are considered to be reasonably efficient feeders, assimilating approximately 80% of the food that is ingested (Bloomberg 1981; Peck et al. 1987; Yamasaki 1998). When in culture H. iris are fed at a daily rate equivalent to  $\sim 3\%$  of their total body weight (Beatie 1998).

No robust studies could be found that describe actual environmental effects from culturing paua in sea-based containment systems. The two known farms in New Zealand are small in both scale and intensity, and function in a co-culture situation which makes species-specific assessments difficult. Some nutrient (ammonia and nitrate) monitoring is required by resource

consent around at least one of these farms and we are advised that reported concentrations have been negligible. It is also worth pointing out that abalone are themselves considered to be highly sensitive to physical and chemical changes in their environment, and thereby demand maintenance of very high water quality conditions within the barrels. Wider effects to the water column are therefore inherently counter-productive to paua farming.

The 'potential' effects of farming paua have however been considered in the past for consent purposes (McShane 1997), and that report identified significant deposition and benthic enrichment issues. However, these findings were apparently based on incorrect feeding and stocking values, which were detrimental to the assessment (Beatie 1998). An alternative theoretical scenario is provided in Table 6 based on published assimilation rates and farming statistics detailed in Beatie (1998) to help gauge the relative potential for benthic effects. This relatively crude calculation suggests that a paua farm could conceivably produce waste products at a rate of  $\sim 2-20$  kg m<sup>2</sup> vr<sup>-1</sup> (dependent on animal size). The upper range of this estimate is comparable to that of modelled predictions for functioning salmon farms in the Marlborough Sounds (Keeley et al. 2006). It is therefore fair to assume that given sufficient scale and commercial intensity, waste production may be sufficient to induce deposition and enrichment related effects. As with other forms of aquaculture, the extent of these effects will be influenced by environmental and farming management practices (e.g. Section 2.3.5, Forrest et al. 2007), which need to be considered in an overall site assessment. It is possible for example, that such commercial intensities are impractical due to the water quality feedback mechanism discussed above.

Table 6.Parameters used in hypothetical scenario for calculating depositional output from a sea-based paua<br/>farm. (Stocking densities and feed rates from Beatie (1998), weight of individuals from McShane<br/>(1997), and feed assimilation rates average from Bloomberg (1981), Peck *et al.* (1987) and<br/>Yamasaki (1998)).

	Animal size		
Parameter	75mm	15mm	Units
Equivalent seabed density	127	952	ind·m <sup>2</sup>
Size (individual)	75	15	mm
Weight (individual)	72	1	g∙ind <sup>-1</sup>
Total biomass	9144	952	$g \cdot m^2$
Feed rate	3	3	% body weight·d <sup>-1</sup>
Food assimilation rate	80	80	%
			2
Feed in	274.3	28.6	$g \cdot m^2 \cdot d$
Organic matter egested as faeces/	54.9	5.7	g⋅m <sup>2</sup> ⋅d
pseudofaeces = Depositional rate	20.0	2.1	kg∙m <sup>2</sup> •yr

Paua are broadcast spawners, which means gametes from adult farmed stock may mix with those of wild populations and farmed progeny may settle on adjacent natural coastline given suitable habitat. As long as good brood stock management protocols are adhered to, risks of genetic contamination issues arsing are likely to be small.



In terms of pathology, abalone are susceptible to a number of disorders, diseases, viruses and parasites (see Appendix 2), none of which are presently problematic in New Zealand paua. Of particular relevance, however, is a Tasmanian example where a farm-originated virus spread to wild abalone resulting in significant mortalities (Hine 2006). Good stock management and biosecurity practices and surveillance are obviously required with this species to help manage pathological threats. Specific mitigation actions should be devised from best practice elsewhere if diseases become established in New Zealand. A factor in our favour is that some parasites appear to have an indirect life cycle requiring a number of host stages. If any of these have been left behind and there are no local substitutes then the pathogen may not be sustainable. This possibility, while encouraging, is no basis for complacency and further work is required to allow more informed inferences.

#### Sea cucumber (Stichopus mollis)

Sea cucumbers are deposit-feeders, obtaining their nutritional requirements from processing large volumes of sediments on the seafloor, digesting the organic components (algae, diatoms, cyanobacteria) and excreting unwanted sediments (Uthicke 1999). They, therefore, require a sediment substrate and are not conducive to suspended culture. Recent experiments have used bottom-oriented cages (Slater & Carton 2007) with some degree of success and *S. mollis* has serious potential as a future aquaculture species. Presently however, juveniles would have to be hatchery spawned because *S. mollis* is not currently listed as harvestable spat under the Fisheries Act (1996).

Examples of sea cucumbers being cultured in isolation are rare, as are any studies relating to adverse environmental effects that can arise from their culture. Instead, studies of environmental effects associated with sea cucumbers tend to focus on their ability to mitigate the depositional effects from the culturing of other species. Hence, sea cucumbers are becoming a popular co-culture species with bivalve (*e.g.* oysters, Paltzat *et al.* 2008), paua (Kang *et al.* 2003) and fish farms (Ahlgren 1998). Sea cucumbers are presently not cultured commercially in New Zealand, but *S. mollis* is being investigated as a potential co-culture species with Greenshell<sup>TM</sup> mussels (Slater & Carton 2007). The mitigative potential of co-culture systems are discussed in more detail in Section 6.1.2. Other potential issues associated with culturing *S. mollis* remain undescribed and are unlikely to be realised until the species is cultured in significant quantities.

#### Kina (Evechinus chloroticus)

Aquaculture of kina in New Zealand is presently restricted to experimental research, and as such, studies relating to environmental effects from commercial-scale culture do not exist. Kina (also known as sea eggs) are sea urchins and belong to a large group of marine invertebrates called Echinoderms and are endemic to New Zealand. Aquaculture trials to date have utilised sea- and land-based cages, in which the animals are fed natural kelp, or specially formulated artificial feeds (James 2006). Much of the research (nationally and internationally) conducted to date has been oriented around factors which affect sea urchin roe enhancement (Klinger *et al.* 1997; James & Heath 2008; Woods *et al.* 2008). The diet that has been trialled most recently in New Zealand, by NIWA, is comprised mostly of protein-rich fish-skins, a

fisheries by-product (Woods *et al.* 2008). In Scotland, urchins have also been used experimentally in polyculture systems beneath salmon farms where they were maintained successfully on excess fish pellets (Kelly *et al.* 1998). We note however, that most of the trials cited in this review were conducted on urchins sourced from the wild. The emphasis appears to still be on conditioning roe ready for market (*i.e.* finishing diets) and as such, the sea-based aquaculture component of this industry may be limited.

Environmental effects associated with kina culture are likely to be minimal as long as the scale and intensity are moderate, and probably akin to those of sea-based paua farming. There is also recognised potential for farming kina as a mitigative component in integrated aquaculture systems. Some potential for contamination issues exists if they are predominantly fed on artificial diets.

## 4.3.4. Scavengers and piscivores

This group is broadly united by the requirement of high protein diets derived from animals and animal by-products (*e.g.* fish, shrimps (brine), fish meal, shellfish *etc.*). Potential culture species included in this group are crayfish (*Jasus edwardsii* and *Sagmariasus verreauxi*), shrimp (various), seahorses (*Hippocampus abdominalis*), and paddle crabs (*Ovalipes catharus*). Culture of these species in New Zealand, is either in its infancy, or likely to occur in land-based systems and are therefore not encompassed by this review. The main exception to this is crayfish, which may be grown in sea-based cages, and have considerably more economic potential due to a consistently high market value.

## Crayfish (J. edwardsii)

Increasing global demand, high economic value and concerns about the sustainability of the wild fisheries have ensured strong world-wide interest in developing culture techniques for crayfish or 'lobsters'. In New Zealand (and Australia), efforts have been focused on the primary wild-caught species, *Jasus edwardsii*. Commercial production has apparently been achieved (Sheppard *et al.* 2002), but hatchery production (from larvae) is still not feasible due to a long and complex larval development phase (Kittaka *et al.* 2005; Williams 2007). Instead, emphasis has been on on-growing wild-caught juveniles or 'pueruli' (Mills & Crear 2004). Commercial culture is, therefore, dependant on the effectiveness and reliability of capturing wild puerulus, and any large scale operations would likely have implications for the sustainability for the wild fishery. In the past this has been addressed by off-setting a set number of puerulus obtained from the wild against tonne of crayfish quota and requiring that a certain percentage of those reared are returned to the quota area from which the pueruli were obtained.

Not surprisingly, studies which specifically deal with environmental effects associated with farming crayfish are scarce. Our review of effects is, therefore, limited to an assessment of potential for inducing effects based on what is known about the likely culture methods. Defined as 'opportunistic carnivores', *J. edwardsii* can feed on a wide variety of invertebrates (Williams 2007), but in culture, can be sustained solely on mussels. Recently however, much
effort has gone into formulating pelleted diets to optimise growth (Simon & James 2007; Williams 2007). As with other forms of aquaculture, it is these high-protein external feed inputs and the subsequent waste products that increase the scope for enrichment-related effects to the seabed and water column. Various cage configurations have been proposed, but growers generally conform to using surface-oriented structures with suspended cages that occupy the top 2-3 m of the water column and in a recent trial were stocked with 35 individuals/m<sup>2</sup>. Enrichment-related effects will obviously also be scale and stocking-density dependent.

Although the gregarious nature of spiny lobsters coupled with access to abundant prey may make them robust to the pressures of high population density (Behringer & Butler 2006), disease outbreaks and mortality within caged populations can be an issue (Diggles 2001). It would therefore follow that the potential for disease transfer to wild populations is an issue worthy of consideration. In terms of potential to influence the genetics of wild fish from culture and transfer of farmed fish, the level of genetic structuring of *J. edwardsii* is thought to be relatively low due to the long planktonic larval phase, which potentially provides for extensive movement and gene flow. This occurs to the extent that there is thought to be some trans-Tasman larval flow (Chiswell *et al.* 2003) and that likely leads to the New Zealand and Australian populations being genetically indistinguishable (Ovenden *et al.* 1999). The risk of crayfish aquaculture influencing natural genetic profiles is therefore presently assessed to be low. It is possible however, that a more rigorous appraisal using new technology may well reveal something different (J. Gardner, pers. comm.).

#### 4.3.5. Macroalgae (seaweeds)

Macroalgae derive their food requirements from dissolved nutrients and sunlight. A broad range of species are cultured worldwide for human consumption (*e.g.* nori, *Porphyra* spp.; wakame, *Undaria pinnatifida*; kombu, *Laminaria japonica*; phycocolloides), food products (e.g. agar, carrageen and alginates), pharmaceutical products and for use in agricultural feeds (Smit 2004). Despite the ~150 farms that have permits to culture seaweed (MFish, pers. comm.), the only species presently being utilised in New Zealand are the large brown algae *Macrocystis pyrifera* and the introduced brown algae *U. pinnatifida*. *M. pyrifera* is predominantly harvested from the wild, but also from an established AMA on Banks Peninsula. The volumes of *M. pyrifera* being harvested are small and it colonises the structures naturally, so can only be loosely described as aquaculture.

Likewise, *U. pinnatifida* is presently a "by-product" of the mussel industry, as it grows profusely on the upper parts of mussel lines. The aquaculture potential of *U. pinnatifida* has been researched by the Cawthron Institute (Hay & Gibbs 1996; Gibbs *et al.* 1998, 2000), but it is yet to be realised because it has been classed as an unwanted organism under the Biosecurity Act 1995. *U. pinnatifida* is also not in the QMS, and therefore has no TAC or TACC. Despite this, one marine farm in Mahanga Bay has been granted permission for *U. pinnatifida* aquaculture, and at least two companies now hold licenses to harvest the species. The estimated tonnage harvested from one of these sites (in the Marlborough Sounds) is 2,000 tonnes year<sup>-1</sup>.

The environmental effects of algae culture in New Zealand remain undetermined due to the absence of commercial scale examples. Internationally, studies pertaining to adverse environmental effects from farming algae are also sparse. There is, however, a wealth of new literature considering the bioremediation potential of culturing algae in integrated systems and its ability mop up excess nutrients discharged from fish farms (Zhou *et al.* 2006a; Blouin *et al.* 2007; Kang *et al.* 2008; Xu *et al.* 2008). While macroalgae farming may be appropriate in a eutrophic system, or in conjunction with an artificial nutrient source, the high nutritional requirements could potentially affect the wider ecosystem in areas that are nutrient poor.

The light requirement of algae culture is likely to impose depth constraints on the culture methods in most situations. This in turn may create a tendency to densely occupy space on the horizontal plane at the surface, which would reduce the amount of light penetration lower in the water column and at the seabed, potentially resulting in localised primary productivity issues. Such effects have been identified from intertidal algae farms, which can impede growth in adjacent seagrass beds and alter the macrofauna community contained within (Eklöf *et al.* 2005). A subsequent study also demonstrated that algae farms can alter the composition of fish communities and potentially increase fish catches (Eklöf *et al.* 2005, Eklöf *et al.* 2006). These findings are however very much site-, situation- and species-specific. Algae farming in New Zealand is more likely to be conducted off the bottom, in deeper water, and as such, the scope for analogous effects will be reduced.

Perhaps the biggest potential environmental issue associated with algae culture relates to biosecurity, as there are numerous examples where introduced species of macroalgae have posed major threats to the surrounding ecosystem, and in some cases, other forms of aquaculture (Forrest *et al.* 2000; Neil *et al.* 2006; Schaffelke *et al.* 2006; Bullard *et al.* 2007). However, these threats are negligible if algae aquaculture is restricted to indigenous species.



## 5. EVALUATION OF ECOLOGICAL RISKS FROM NON-FINFISH AQUACULTURE

#### 5.1. Frame work for risk assessment

Our review highlights several commonalities with regard to ecological effects associated with the subtidal cultivation of Greenshell<sup>TM</sup> mussels and intertidal cultivation of Pacific oysters, particularly with regard to seabed and water column effects. Farming of minor or potential species of filter-feeding bivalves (scallops, flat oysters, subtidal Pacific oysters, blue mussels) is expected to have analogous effects on the marine environment, whereas the cultivation of organisms that require the addition of feed (paua, crayfish) may lead to different types or magnitudes of effects. The nature and magnitude of wider ecological effects such as the spread of pest species, disease outbreaks, or effects on the genetic makeup of natural populations will often depend on the species.

From the available information, and based on experience with other forms of aquaculture or from wider ecological literature, we consider that the key ecological stressors that lead to adverse effects are sufficiently recognised that they can be evaluated across the various forms of non-finfish aquaculture. Such an evaluation is useful in that it places the effects described in this report within a comparative "risk" context, which considers the magnitude, likelihood, spatial extent and duration (impact persistence over time) of an effect. Such an evaluation can also assist in identifying knowledge gaps and guiding management or mitigation strategies. Rather than attempt such an evaluation for all actual or potential aquaculture species described in this report, we provide by way of an example a summary of the analysis by Forrest et al. (2009) for intertidal Pacific oyster farming in New Zealand. That study evaluated the relative ecological significance of the range of issues discussed in this report, with the exception of marine mammals. Effects to marine mammals are not well evaluated by this type of risk frame work, as the likelihood of any adverse effect is very small (in fact we are aware no adverse interactions in relation to oyster cultivation in New Zealand) and the consequences highly species- and situation-specific. For example, the death of a single individual from a population of abundant and common animals (e.g. New Zealand fur seal) may have relatively limited significance. However, if that individual was a large reproductive adult from a small population of an endangered species, then the consequences could be profound.

Hence, except for marine mammal interactions, oyster cultivation risk for the other issues is evaluated for current levels of development in relation to three categories: (i) the magnitude of effects, including the likelihood and consequences of actual or potential effects; (ii) the spatial extent of effects from site-specific to regional scales; and (iii) duration; the length of time effects would persist if farming operations were ceased and farm structures removed. Within each category, ecological significance and uncertainty were ranked according to narrative criteria (Table 7). This exercise was undertaken by seven Cawthron scientists with a broad knowledge of the ecological effects of marine aquaculture.

Category	Low	Medium	High
Level of knowledge	Based on perception or inference from related studies	Based on limited information on effects of elevated culture	Specific effects of elevated culture known
Relative magnitude of effect	Minor	Moderate	Significant
Spatial extent of effect	Local scale (restricted to tens of metres from culture area)	Bay-wide (extending hundreds of metres from culture area)	Regional (kilometres or more from culture area)
Duration of effect	Short-term (abates immediately)	Medium-term (continues for months to a few years)	Long-term (continues for many years and may be irreversible)

# **Table 7.**Narrative criteria used to compare relative level of knowledge and ecological significance of<br/>effects from elevated oyster culture.

The results of this evaluation (Table 8) **should be regarded as a guide only**, as they are intended to reflect *relative* risk (as derived from expert opinion) and in some cases are based on limited information. Furthermore, actual levels of risk will often depend on site-specific factors such as the intensity of farming in a given area, the sensitivity of the receiving environment, the presence of pre-existing stressors, and the extent to which mitigation is possible. Although we suggest that only major differences in risk scores are meaningful, the evaluation nonetheless facilitates general understanding of the ecological significance of the various issues at least in a relative sense.

### 5.2. Risk evaluation

Results of this exercise revealed that biosecurity issues relating to the spread of pest organisms received the highest risk scores (Table 8). This finding is consistent with an aquaculture risk assessment described by Crawford (2003) for Tasmania, and also with the general view that inadvertent pest introductions are one of the more significant issues associated with aquaculture in estuaries (deFur & Rader 2003). The reason is that, by comparison with all other risk categories, the spread of pest organisms by aquaculture activities can occur at regional scales, potentially leading to ecologically significant and irreversible changes to coastal ecosystems (Elliot 2003). Whether the spread of a given pest organism (or oysters themselves) by oyster farming activities (*e.g.* inter-estuary transfers of infected equipment or seed-stock) is conisdered significant depends on a number of different factors (*e.g.* relative risks from other pathways such as natural dispersal or fouled vessels). Furthermore, it is important to recognise that management plans and mitigation strategies may be developed to minimise biosecurity risks.

Seabed effects from biodeposition and debris, and potential effects of disease received the next highest relative risk scores. The effects of biodeposition and debris are the more obvious or conspicuous effects of oyster farms, and are reasonably well understood. In general, seabed



effects can be moderately pronounced but highly site-specific, appear to extend no more than a few tens of metres from the perimeter of the farmed area at worst, and are likely to be reversible (should farming be discontinued) over time scales of several months to a few years, although debris accumulation could lead to long-term changes in habitat structure. Thus the wider ecosystem significance of seabed effects depends on the spatial scale of farming activity in relation to site-specific ecological values, such as the presence of species or habitats that are sensitive to impacts or are of special interest (*e.g.* high conservation values, keystone species). The potential for disease also scored relatively high; even though the likelihood was considered low the consequences could be high (hence magnitude medium), and include the potential for widespread and long-term effects.

**Table 8.**Summary of actual and potential effects from elevated intertidal oyster culture in relation to key<br/>risk criteria fromTable 7. Note that marine mammals were excluded from this assessment for<br/>reasons discussed in the text above.

Effects	Description	Knowledge	Magnitude	Likely spatial extent	Likely duration <sup>a</sup>	Relative risk rank
BENTHIC						
Biodeposition and enrichment	Altered sediments and enrichment of benthos	High	Low-medium	Lowl	Medium	Medium
Effects of debris and altered topography	Structural change to seabed habitat and effects on epibenthic communities	Medium	Medium	Low	High <sup>b</sup>	Medium
Physical disturbance	Seabed disturbance from farm operations	Medium	Medium	Low	Medium	Low- medium
Shading	Effects on benthic primary producers if present	Medium	Low- medium	Low	Medium	Low- medium
Contaminant inputs	Contaminant accumulation and ecotoxic effects	Low	Low	Low	Medium	Low
WATER COLUMN						
Alteration to waves and currents	Reduced flushing and increased sedimentation	Medium	Low- medium	Low	Low	Low
Adverse effects on water quality	Dissolved oxygen depletion and sulphide production	Medium	Low	Low	Medium <sup>c</sup>	Low
Nutrient cycling and seston depletion	Alteration to nutrient cycles and depletion of seston	Low-medium	Low- medium	Medium	Uncertain	Low- medium
OTHER						
Habitat creation	New habitat provided by farm structures	Low-medium	Low- medium	Medium	Uncertain <sup>d</sup>	Low- medium
Fish	Alteration to food sources, provision of novel habitat	Low-medium	Low- medium	Medium	Uncertain <sup>d</sup>	Low- medium
Seabirds	Alteration to food sources and foraging habitat, provision of novel habitat	Low-medium	Low- medium	Medium	Uncertain <sup>d</sup>	Low- medium
Pest species	Introduction and spread of	Medium-high	High	High	High <sup>b</sup>	High



Dise

	fouling pests and toxic or noxious microalgae					
ise	Introduction and spread of parasites or pathogens	Medium	Low	High	High <sup>b</sup>	Medium

<sup>a</sup> Duration of effect assumes farming ceases and structures are removed

<sup>b</sup> Potential to mitigate the potential for long-term effects by managing risk

<sup>c</sup> Time-scale of recovery from adverse water quality effects may be linked to sediment recovery

<sup>d</sup> Uncertain; insufficient knowledge of effect duration to make assessment

There are a range of remaining issues relating mainly to water column and wider ecosystem effects for which ecological significance was on average scored as low or low-medium in a relative sense (Table 8). This generally concurs with the review of Dumbauld et al. (2009) who concluded that bivalve culture effects in US West Coast estuaries tended to be localised and short-term, and not associated with larger scale ecosystem changes. Feedback from the experts in the present work revealed that, even where the level of knowledge or certainty was regarded as low, relatively low scores were often assigned on the basis that effects had not been conclusively documented (*e.g.* seaston/food depletion), or were not judged as adverse (e.g. novel habitat creation). Clearly, in the latter case where this viewpoint is highly subjective, there is potential for greatly differing views, although in this instance the general consistency in views for most categories was reflected in a low variance around mean risk and knowledge/certainty scores (see Forrest et al. 2009). We also acknowledge the possibility that unrecognised estuary-wide or cumulative effects could have already occurred from some oyster farm developments, or could arise in the future, for example: (i) in situations of highintensity oyster farming (e.g. if there are enclosed embayments dominated by oyster farms), or (ii) because of the occurrence of ecological values of high importance. Without knowledge of baseline conditions and subsequent changes post-farm development, many of the water column effects and wider ecosystem effects described in this report would be difficult to determine retrospectively.

The results from the risk assessment of oyster farming would be broadly transferable to the subtidal cultivation of Greenshell<sup>TM</sup> mussels and other bivalve species. Perhaps the main point of difference would be in relation to seabed effects and changes in topography, which are different for intertidal versus subtidal cultivation, due in large part to the proximity of the structures to the seabed and the water depth. The effects of the farm structures on currents and shading would also vary depending on the type of structures used (see Figure 22). The magnitude, spatial extent and duration of effects arising from the spread of pest species and/or disease is considered high for all cultured non-finfish species; however, the pests and/or diseases involved are likely specific to the type of cultivation method and species. As discussed in Section 4.3.4, the farming of organisms such as crayfish and paua that require the addition of food to the marine environment would potentially lead to differences in the effects on the seabed and water column. For instance, the effects of finfish farming, which involves addition of large amounts of feed, can result in more pronounced seabed effects than those associated with oyster or mussel farming, which has less of an enrichment effect on the benthos, but potentially a greater effect on the water column through seston depletion.



Farming of macroalgae or deposit feeders such as sea cucumbers would be expected to pose less risk to the environment than the cultivation of shellfish. If co-cultured with shellfish or finfish, they could assist in mitigating adverse effects on the seabed and water column.

While the notion of ecological risk tends to imply a negative or adverse effect, there are clearly some ecological effects from farming shellfish that could subjectively be considered as 'beneficial'. Furthermore, when the range of effects is considered as a whole it could be argued that some nominally 'adverse' effects may be compensated to some extent by more 'positive' effects. For example, although natural seabed sediments and benthos may be altered beneath oyster and mussel farms, local biodiversity and production may be enhanced through provision of habitat for fouling. Hence this range of 'beneficial' to 'adverse' effects needs to be balanced against each other when considering aquaculture developments. Even more broadly, we suggest that management responses to farm developments be made in relation to other sources of environmental risk to estuarine systems at a bay-wide or regional scale, so that the effects of aquaculture are placed in context. This approach was recently applied to mussel farm development in New Zealand (Elmetri & Felsing 2006) using a Relative Risk Model approach described by Landis (2004) and Landis & Wiegers (1997). In this approach, the relative risk to predefined endpoints (e.g. particular species, populations, habitats) from a number of sources and stressors in addition to oyster cultivation (e.g. agricultural land use, urban development, fishing, climate change) could be investigated and assessed. Such methods can be applied in defined regions (e.g. estuaries) or across multiple regions, providing a basis for developing plans for research and prioritising management according to the greatest sources of risk.



## 6. FUTURE DEVELOPMENTS IN NEW ZEALAND AQUACULTURE

The amount of space designated for aquaculture has expanded dramatically in the past ten years. This is in accordance with the Food and Beverage Taskforce commitment to help the aquaculture sector achieve its goal of \$1 billion in sales per annum by 2025. The net result could conceivably be a four-fold increase in farmable space (excluding scallop enhancement areas) over a ten year period, from 4515 ha at the beginning of 2001 to ~20,000 ha by 2011; most of which will be arranged in large individual AMAs. The scale of these sites is such that they are likely to spawn a suite of new innovations and management practices, from on-site farming techniques, to processing and marketing, and with that, conceivably new scale-related environmental effects. The nature of such changes will become apparent over the next five to ten years as these larger AMAs become fully utilised. It is worth noting however, that many of the large AMAs are restricted to staged development, with the progression between stages being dependent on there being no adverse environmental effects.

In terms of the major existing grow-out systems, technologies for mussels are well established and unlikely to change dramatically in the near future. Within the Pacific oyster industry, there is currently a shift away from traditional stick grow-out towards bag or basket grow-out systems (*e.g.* BST grow-out system). While these systems require a higher capital investment, they produce a higher value product and enable greater control over the grow-out process. It is likely that within the next five years, there will be a further development with systems developed for integrating Pacific oyster culture with existing mussel long-line technology. The high value of Pacific oysters (compared with mussels) provides the incentive for this species substitution.

A key development within the mussel industry over the next five years is likely to revolve around access to hatchery spat. This will remove the reliance on existing wild spat catch, potentially reduce the proportion of farm area required for spat management, give better control of harvest timing, and allow the deployment of selectively bred mussels. Selective breeding is likely to focus initially on increasing productivity (with both faster growing mussels and mussels that utilise available food resources more efficiently), but a shift to adding value is likely to occur as suitable traits for improvement are identified. This has been the case with Pacific oyster breeding in New Zealand, where sufficient growth rate gains have been achieved and the focus has shifted to improving product quality. With increased dependency on hatchery spat comes increased potential for genetic issues and this will need to be considered carefully as the industry develops.

The integration of Pacific oysters with mussel farming operations is symptomatic of a general interest in growing new high value shellfish species using the existing farming infrastructure. Cawthron is working with Aquaculture New Zealand to identify and domesticate suitable species for development as emerging aquaculture species. Potential candidates include the flat (Bluff or Tasman Bay) oyster and scallop. Both are high-value filter-feeding species (like mussels and Pacific oysters) where the technical barriers to domestication are likely to be



resolved within a 5-10 year horizon. With sufficient market pull, the large-scale production of these species could be achieved relatively quickly without requiring significant capital investment, by integrating with existing production infrastructure.

Research and/or growth trials are being carried out with a number of other marine species including scallops (*P. novaezelandiae*), sea cucumber (*Stichopus mollis*), various macroalgae, rock lobster (*Jasus edwardsii*), sponges, and seahorses. It is possible that some of these species will be produced in commercial quantities within a five to ten year time frame, however, they are likely to remain minor species (in terms of biomass at least) for some time. Although not considered within this report, several finfish species (such as groper, kingfish, snapper and green bone) are also reportedly close to commercialisation and indeed finfish farming is likely to become an increasingly important aquaculture sector. The implications of culturing more than one of these species within the same farm area are discussed in Section 6.1.2.

#### 6.1.1. Offshore aquaculture

As described above, much of the new aquaculture space that has been created in New Zealand is situated offshore, away from the protection of land and in generally more challenging environments. While moving offshore is viewed as a compromise by some in the industry, it has facilitated the desired expansion, and now represents the new frontier of marine aquaculture. Feasibility and optimum utilisation of this space is presently being determined as part of a Government FRST-funded research project (CAWX0302) based around the Hawke Bay open-ocean site. As part of this research, New Zealand's first commercial offshore line has been installed and maintained for ~3 years, and a range of different shellfish species have been trialled with some positive results.

From an environmental perspective, the open ocean represents a much larger system with greater resilience to effects and the necessary spatial scope to de-intensify some traditional farming practices. Although poorly understood at present, the role waves play in the assimilation, resuspension and dispersal of bio-deposits is likely to be particularly important in terms of benthic and water column impact mitigation. By their very nature, open-ocean sites also tend to be situated over soft-sediment habitats that are traditionally only mildly impacted by mussel farming activities (Section 2.3.5). Indeed, preliminary monitoring results from studies beneath New Zealand's first offshore mussel line have so far described a natural, unaffected benthos (Keeley 2006; Sneddon & Keeley 2007).

Presently however, assessments of environmental issues resulting from full-scale operation is limited to predictions, based on: 1) our understanding of the un-altered offshore environment, 2) experience farming in-shore, 3) a small literature pertaining to other semi-analogous offshore structures (*i.e.* oil platforms and FAD's), and 4) ecological intuition. Potential environmental issues associated with farming finfish offshore were reviewed in this light and evaluated in terms of overall risk in Keeley *et al.* (2007). The main findings that were pertinent to farming of 'other species' offshore, are discussed below.



Figure 23 "Innovator" installing New Zealand's first offshore mussel line at Hawke Bay AMA.

#### **Risk evaluation**

Areas of environmental compliance that are considered to pose the most risk in the offshore setting, included: seabed effects, biosecurity, habitat creations and wild fish, escapees and genetic contamination, disease and parasites and effects to marine mammals. Potential for seabed effects ranked high primarily because the knowledge and certainty around the issues is considered high, and although it may be effectively managed by reducing stock densities and feeding rates, this is unlikely to be a commercially viable option; therefore, manageability is limited. This issue mainly applies to farming of species that require additional feeds, such as finfish, crayfish, kina and paua. As described throughout this document, extractive forms of aquaculture (*i.e.* filter-feeding organisms) do not have the same scope for benthic effects, particularly in more dissipative and expansive offshore environments. Biosecurity still ranked highly because 1) the effects are potentially irreversible (Forrest et al. 2007), 2) the likelihood of issues arising increase with a large-scale farm, and 3) the control of all possible vectors is impractical (*i.e.* limited manageability). This is particularly pertinent to offshore sites, because they tend to become a focal point for recreational fishers, as has been observed at Hawke Bay and Opotiki (N Keeley, pers. obs.). However, most recreational vessels are trailerable and therefore less prone to fouling issues. It is also possible that the spatial isolation of offshore sites removes them from inshore propagation pressure, but the extent to which this occurs is unclear and the subject of on-going FRST funded research.

Genetic effects from escapees were ranked highly for finfish culture species that are endemic to the area, due to the added difficulties of maintaining cage integrity. While this may apply to other mobile species, such as crayfish, it does not necessarily follow for sessile organisms such as mussels. Also of importance here are factors such as the distance of the farm from viable habitat and the dispersal range of gametes from the species concerned; hence it is very much a



species-specific issue. The same is true for the management of disease and parasites. Habitat alteration and the associated effects to wild fish populations represents a reasonable risk because farm structures will inevitably be colonised by communities that are not otherwise found in an offshore environment (*i.e.* deep, featureless mud bottom). The effects that this new habitat, in conjunction with an additional artificial food source, may have on wild fish could be important. However, the possible 'severity' of effects was considered low because most wild fish populations are more likely to be 'positively', rather than detrimentally affected. As described in Section 2.5.2, there remains a reasonable degree of uncertainty around this issue.

In terms of risks to marine mammals, the difficulties associated with ranking them using this kind of framework (as discussed in section 5.2) also apply for the offshore setting. Nevertheless, in our assessment, risks to marine mammals from offshore aquaculture are slightly greater than for inshore sites due to the increased likelihood that the activity may be situated near to, or within, important marine mammal habitat. It is also worth noting however, that threats to marine mammals mainly arise from loose ropes (section 2.5.4) and therefore, may be minimised through implementing appropriate 'best management practices'.

Issues that were considered to pose a lower environmental risk include: effects to localised primary production, chemical contamination, effects to seabirds, harmful algal bloom incidence and magnitude and dissolved oxygen depletion. Primary production, or eutrophication, effects were assessed to be the most important of these five, primarily because of the scope for flow-on ecosystem effects. The latter four of these issues scored low in terms of both likelihood of occurrence and potential for negative environmental consequence. Effects to seabirds were interesting in this regard, as it was considered very relevant to offshore sites with the potential to significantly alter bird distributions. However, bird numbers were considered more likely to increase than decrease in farmed locations and the associated consequences were minimal (Keeley *et al.* 2007).

#### Conclusion

New Zealand and indeed, the world, are yet to experience a fully operational mussel farm of the scale that is planned for some of the existing offshore sites. While many environmental indicators point towards an increased resilience to typical enrichment-related effects, it is also conceivable that new scale-related issues will arise. For example, the cumulative effect of a vast area of structures on hydrodynamics and associated biological processes is difficult to predict and remains undetermined (Goodwin *et al.* 2008; Stevens *et al.* 2008). Economics will inevitably dictate that even large spaces are optimally utilised, which means one or more forms of carrying capacity (see Section 2.4.4) may eventually be reached. Hence, the need for staged development coupled with careful monitoring and pre-established adaptive management responses. There may also be scale-related thresholds that influence the establishment, development and subsequent composition of fouling organisms that will only be realised through experience. It is also conceivable however, that some of the effects will be positive, such as increased abundances of wild fish from the creation of complex pseudo-reef habitat and the likely alleviation of exploitation pressure on commercial fish species. Moreover,



larger AMAs (*e.g.* >500 ha) may provide the necessary space for designing functional integrated aquaculture systems, which have the potential to mitigate adverse effects from nutrient rich farm wastes (Chopin *et al.* 2001; Xu 2008; see below - Section 6.1.2).

#### 6.1.2. Integrated culture systems

Internationally, developing integrated culture systems (a.k.a. 'polyculture' or 'co-culture'), has been touted as an important future direction for optimisation and sustainability of aquaculture. Growing selected organisms, usually of different trophic levels, side by side can provide nutrient bioremediation capability, mutual benefits to the co-cultured organisms, economic diversification by producing other value-added crops and increased profitability per cultivation unit for the industry as a whole (Chopin et al. 2001). Although there have been mixed results with experiments worldwide, the positive literature with respect to productivity (Stirling & Okumus 1995; Kang et al. 2003; Zenone & Sarà 2007; Paltzat et al. 2008), and impact mitigation (Zhou et al. 2006; Slater & Carton 2007; Hayashi et al. 2008; Kang et al. 2008), is growing. However, much of the work conducted to date remains in the realms of research, and the commercial uptake of systems has been slow due to a number of scale-related uncertainties (see Troell et al. 2003). There are also some possible drawbacks to co-culture of different species. For example, some studies have identified that mussels can act as reservoirs of bacteria pathogenic to fish (Stirling & Okumuş 1995; Kitamura et al. 2007; see Appendix 2). The relevance of this issue to New Zealand species is discussed in the respective disease section of this report (*i.e.* 2.5.6, 3.5.6, Appendix 2), but in most instances, remains largely undetermined. In addition, although not quantitatively assessed, combinations of different organisms and associated structures has the potential to create flow impedance issues, thereby affecting flushing rates and related environmental issues.

The potential combinations of New Zealand species are many and varied. Combinations with particular scope for mitigation of environmental effects include: growing sea cucumbers beneath shellfish farms to process deposited organic material (Slater & Carton 2007; Paltzat et al. 2008), culturing shellfish around finfish farms to intercept organic particulates and dissolved nutrients (Jones & Iwama 1991, Stirling & Okumuş 1995; Lefebvre et al. 2000; La Rosa 2002; Cheshuck et al. 2003), the use of algae around fish and conceivably paua/crayfish/kina farms to utilise excess dissolved nutrients (Chopin et al. 2001; Kang et al. 2003; Zhou et al. 2006; Langdon et al. 2004; Xu et al. 2008; Hayashi et al. 2008; Kang et al. 2008). Although seeming effective for mitigation, the added complexity inherent in co-culture systems means that they are not necessarily commercially practical or economically feasible. There also remains the trade-off between possible benefits co-culture systems and the likely physical draw-back of compromising flushing. They are therefore unlikely to be widely adopted in the near future unless necessitated by compliance with environmental legislation or market demands change. In the case of shellfish farming in New Zealand, environmental effects are relatively minor and the evolution of co-culture is more likely to be driven by production and market diversity incentives - which remain less well proven. However, there remains considerable scope for innovative co-culture in conjunction with fish farming (or other food-added forms, e.g. paua, crayfish), where significant enrichment problems can occur



(Forrest *et al.* 2007). There are also signs of a general shift in farming philosophy, toward placing a greater emphasis on sustainability and optimal use of resources, and to this end, co-culture may play a pivotal role.



## 7. MANAGEMENT AND MITIGATION OF ECOLOGICAL EFFECTS

### 7.1. Site selection

Our review highlights that the nature and magnitude of effects largely depend on site-specific conditions relating to the intensity of farming, flushing characteristics of the environment, and the proximity of the farm to valued habitats (*e.g.* rocky reefs) and species (*e.g.* nesting shorebirds). Effects to the seabed may be reduced by locating farms in high current environments or open coastal situations in sufficient depths such that increased currents and wave action enhances dispersion of farm-generated wastes over a wider area. Tools such as predictive depositional models (DEPOMOD; Cromey *et al.* 2000) can be useful in estimating the spatial extent and magnitude of effects prior to new developments. Monitoring data suggests that diffuse organic loading beyond the immediate deposition footprint can be effectively assimilated into the receiving environment and may in fact result in a slight increase in productivity in the sediments (Forrest *et al.* 2007). Wider ecological effects on marine mammals and seabirds can be effectively mitigated through properly placing farms in locations that are well removed from critical breeding and foraging areas.

A shift towards high-flow sites does, however, increase the likelihood that farms are situated within close proximity to traditionally 'higher value' communities. The physical properties of high flow sites tend to coincide with non-depositional reef substrates and coarser, well-flushed sediments (*e.g.* gravel, sand, and shell) that are colonised by diverse and complex reef communities facilitated by the strong currents and good food supply. This is particularly relevant in the placement of food-added culture systems, which require particularly high flows to dissipate waste. Hence, the traditional philosophy of placing marine farms in depositional basins is increasingly being challenged. At present enrichment-related effects on reef biota are relatively poorly documented (Keeley *et al.* 2008). Offshore sites may provide the best of both worlds in this regard, as they are often situated in strong current flows, have the added but undefined dispersive contribution of wave action, and are some distance from coastal reef fringes.

## 7.2. Farm management

Shellfish farmers have adopted an environmental code of practice (ECOP) for managing inputs of debris associated with the development and maintenance of farm structures. Entanglement risks for marine mammals can be minimised by adopting measures such as keeping lines taut, using thicker lines when possible (Kemper *et al.* 2003), and ensuring that farms are well maintained (*e.g.* removal of broken or loose crop lines). These types of design and maintenance features, and operational procedures for lines that minimise entanglement risk, have already been implemented at New Zealand shellfish farms as part of the ECOP.

Seabed effects from individual farms can be managed through the development of environmental criteria and maintaining an appropriate stocking density, which can be integrated into adaptive management plans (AMPs), as has been the approach with salmon farming in the Marlborough Sounds (Forrest *et al.* 2007). Monitoring of environmental effects coupled with either staged or adaptive management is a useful approach in situations where there are uncertainties regarding environmental effects due to either the scale of the development or the proximity to habitats of high perceived ecological value or susceptibility to depositional effects. Seabed enrichment effects from feed-added forms of aquaculture can also be mitigated y using single-point mooring systems, which spread the discharge of a greater area (Goudy *et al.* 2001).

## 7.3. Integrated culture

Integrated culture involves the cultivation of two or more species, usually of different trophic levels, in close proximity to one another. It is a rapidly advancing area of research and has considerable scope for mitigation of environmental effects in the future. Uptake by industry is presently constrained by environmental necessity and/or proven economic incentives. See Section 6.1.2 for more details.

## 7.4. Addressing biosecurity risks and managing pest species

The adverse effects of pest introduction and spread, that can have profound non-local and irreversible consequences, are arguably more significant than the commonly cited seabed effects. Clearly, there is a need to redress the balance of effort in future studies. This could include, for example, site-specific risk profiling for actual and potential pests (*e.g.* assessment of the likelihood that high risk species will establish), estimation of the significance of pest spread by marine farming pathways relative to other sources of risk (*e.g.* recreational vessels), and consideration of the feasibility of management.

In recognition of biosecurity risks from aquaculture operations, regional councils are increasingly stipulating conditions on resource consents that require management of biosecurity risk in some form. Unfortunately, some councils have prescribed conditions that are either ineffective (will not lead to biosecurity benefits because significant risks already exist) or for which compliance (or evaluating compliance) is not feasible. In part, this situation may reflect a lack of knowledge and experience with marine biosecurity issues, and is a problem that should ideally be addressed at a national level so that pragmatic consent requirements are achieved and applied consistently among councils.

Irrespective of regulatory requirements, aquaculture companies tend to be pro-active in developing biosecurity management strategies, as they clearly have a strong incentive to protect their operations from the adverse effects of pest species. Among other things, within the mussel and Pacific oyster industries the various strategies have included development of Biosecurity Management Plans, Codes of Practice for seed-stock transfers, and related

methods to manage biofouling (*e.g.* NZMIC 2001; Taylor *et al.* 2005). Additional ways in which aquaculture companies can contribute to the effective management of biosecurity risks were discussed by Forrest (2007).

There are also a range of emerging tools, methods and knowledge from studies in New Zealand and elsewhere that can assist the industry in their management approaches (Forrest & Blakemore 2006; Forrest *et al.* 2006; Coutts & Forrest 2007; Pannell & Coutts 2007; Piola *et al.* 2008). Although the characteristics of pest species and their environments often limit 'manageability', it is also important to recognise that management success can to a large extent hinge on whether there is sufficient buy-in from other coastal operators and government agencies. Attempts by the aquaculture industry to deal with pests may ultimately be futile if such efforts do not have the support of key stakeholders at a regional and national level; failure to eradicate *Didemnum* from the Marlborough Sounds (an undertaking considered by Coutts & Forrest 2007 to be technically feasible) is a case in point.

### 7.5. Disease

A number of diseases documented overseas would have significant impact on the New Zealand aquaculture industry if they were to become established here (Appendix 2). In some cases they also have the potential to impact on wild conspecifics and conceivably associated fisheries. Good biosecurity practices and surveillance as described above are probably the best ways of managing the threat. Specific mitigation factors should be devised from best practice elsewhere should one of these exotic diseases become established.

### 7.6. Managing genetic diversity

A common thread running through this report is that managing genetic diversity of wild, indigenous conspecifics is an important environmental consideration. In most existing forms of aquaculture this risk is already mitigated to a large extent due to stocks being sourced from inherently genetically diverse wild populations (*e.g.* mussels), or by the species being non-indigenous (*i.e.* in the case of Pacific oysters). But there remain risks associated with upscaling the culture of new species, and/or with increasing dependency on hatchery-reared stock. These risks are manageable through identifying the genetic structuring within the wild population prior to implementation and regulating transfer between regions accordingly, and through careful management of selectively bred stock to ensure adequate genetic diversity. For example, steps can also be taken at the grow-out stage to manage and maintain diversity of farmed stock within farms/bays/regions by setting standards for combinations of families within a bay (or other relevant area based on progeny dispersal range). In some instances, farmers may also adopt triploidy, which theoretically negates genetic contamination issues.



## 8. REMAINING INFORMATION GAPS

Overall, this review highlights that our present knowledge of ecological effects arising from New Zealand's two most widely practiced forms of aquaculture is reasonably good. This is particularly true for the more 'conventional' effects associated with the seabed, and to a lesser degree, water column processes. It is also apparent that the wealth of information and level of knowledge is high when put in the context of effects associated with other coastal activities (*e.g.* sedimentation and smothering effects from dredging, NZ and species specific toxicity levels, effects of human activities on marine mammals). However, through this review we have identified areas where knowledge is lacking or can be improved. Inevitably many of these gaps are associated with the culturing of 'new species'. Noted information gaps include the following (in no particular order):

- There is limited information on the actual rates of sedimentation occurring beneath and adjacent to marine farms. Such information is necessary to validate models used in predicting depositional footprints and for determining the rates of deposition that can be effectively assimilated by the environment (*e.g.* deposition may be occurring at distances well beyond the farm but are not detectable based on monitoring indicators such as organic enrichment of sediments). Along these lines there is little available information on the links between seabed effects and the water column (*e.g.* the influence of organic enrichment of the seabed on water-column nutrient chemistry).
- There is little known about the effects of aquaculture and associated biodeposits on high value reef communities that can be found in close proximity to some farm areas. In particular there is a paucity of information surrounding how taxa such as sponges, hydroids, ascidians *etc.*, as well as mobile reef epibiota (*e.g.* crabs, brittle stars), respond to organic deposits. Some tolerant reef communities are considered useful from an impact amelioration standpoint (Angel & Spanier 2002; Gao *et al.* 2008), while others are likely to be highly enrichment sensitive and it would be useful to know more about how various taxa groups respond. Likewise, effects on adjacent intertidal habitats remain poorly documented.
- This study also identified a notable dearth of information surrounding the effects of marine farms on the wider food web and in particular, wild fish assemblages. Although this has not been a big issue to date, it is apparent that the scope for interactions between commercial fish species (as well as other species including marine mammals) and marine farms will increase with the development of the several new large offshore farms. Scale-related effects from larger farms on habitats and associated ecosystem function are difficult to predict and is likely to be an area of some interest during their development.
- Through water column surveys and application of numerical models, we have a reasonably good understanding of the effects of filter feeding bivalves on seston depletion. However, we know little regarding the effects of bivalve aquaculture on the composition of plankton communities, which in turn may have wider ecological effects on the food web. Included in this information gap is the general lack of research surrounding the potential consumption of larval zooplankton species (e.g. fish, crustaceans) and the subsequent ramifications for their recruitment success.



- Considerable growth in the aquaculture industry as anticipated over the next 15 years (NZAS 2006) will in turn require a better understanding of the wider ecosystem effects of shellfish aquaculture, particularly with regard to the cumulative effects of additional and aquaculture development (along side other anthropogenic stressors) within the context of ecological carrying capacity. Research to address wider ecological issues where information is relatively sparse will require understanding of complex ecosystem processes, many of which occur beyond the immediate environment of the cultivation area (*e.g.* changes to food web pathways). Modelling approaches have been undertaken to evaluate trophic effects from culturing oysters (Leguerrier *et al.* 2004) and mussels (Jiang & Gibbs 2005) and further development of these types of models may assist in forecasting cumulative ecosystem-scale effects.
- The relationship between the environment and the growth of the main New Zealand culture species, which underpins any related ecosystem models, is presently poorly defined. A better understanding of the feeding physiology and energetics of New Zealand's main aquaculture species would greatly improve confidence and reduce variance in model outputs, particularly when it comes making predictions for new environments (*e.g.* offshore).
- In Section 7.4 we highlighted a need for better understanding of biosecurity threats. For example, disease outbreaks and transmission from cultured shellfish, while not currently identified as a major issue, does carry with it a high level of risk. Hence we need to understand more about how increasing aquaculture, or perhaps diversifying cultured species, may in turn increase this risk on the New Zealand environment. A useful step would be to gauge the susceptibility of cultured species by assessing novel disease loads in the same organism growing in foreign waters. Other important information needs that would allow better assessment of disease risk include identification of APX (see Appendix 2) to species level and differentiating it (or otherwise) from the APX in flat oysters. Also, life-cycle studies on *Marteilia* to ascertain the stringency of intermediate host specificity.
- Present stock management practices for Greenshell<sup>TM</sup> mussels do not appear to pose a significant threat to the genetic diversity of wild mussels. However, this is partly afforded through the genetic diversity that is implicit in wild-sourced spat and has the potential to change with the advent of selectively breeding for farms. It would appear that more research is required into the potential genetic implications of this practice, leading to the development of sensible brood-stock management and farming protocols.
- Pacific oysters have been reported to show no evidence of reduced genetic variation (Smith *et al.* 1986). This lack of a founder effect is surprising, but on the positive side it would suggest that the naturally high resistance of this species to many infections has not been degraded by genetic bottlenecks in New Zealand populations. Further research is required to confirm this. Such work would also be an opportunity to ascertain gene flow between naturalised and cultivated New Zealand *C. gigas*. This might also afford insights into potential pathogen flows between populations.



## 9. ACKNOWLEDGEMENTS

The authors would like to thank, first and foremost, MFish for the funding and opportunity to compile this report. Secondly we acknowledge Chris Cornelisen (Cawthron) for his significant reviewing and editing role. Also from Cawthron, we thank Nick King, Norman Ragg and other members of the Glenhaven Aquaculture Group for their comments on the status of aquaculture in New Zealand and on research, development and specific physiology aspects. Advice and assistance was also provided by industry representatives Andy Elliot (Whakatu Inc.) and Dan McCall (Marlborough Mussels Ltd). We also thank Graeme Coates and other members of NZMFA Inc. and RRC Ltd for valuable input and support in seeking approval for use of information contained in FRIA documents and various RRC Ltd reports. Finally, thank you to Kathryn Laurence of Cawthron for her role in formatting the draft document.



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# 11. APPENDICES

# Appendix 1. Depletion of phytoplankton by mussel farms

Given the ability of filter feeding shellfish to filter large volumes of water (up to 15 l/hr/g of soft dry tissue weight, Hawkins *et al.* 1999) and the high concentrations shellfish within farm sites, models have hypothesised that local depletion of phytoplankton in the water column should be detectable (e.g. Duarte *et al.* 2008 cites five examples). This depletion may have local impacts on other organisms that utilise phytoplankton or may indicate secondary impacts (*e.g.* zooplankton larval grazing – Gibbs 2004). Effects of aquaculture on phytoplankton depletion may have a cumulative effect in areas of multiple mussel farms. As a consequence of this, chlorophyll surveys have been introduced into fisheries research assessments (FRIAs) to assess the potential for cumulative impacts of new developments on this issue.

These surveys were conducted by pumping water from 2-3 metres depth through a fluorometer to measure chlorophyll to derive spatial estimates of phytoplankton abundance and see if significant depletion was detectable. Although these surveys were undertaken as quickly as possible in order to try to produce a "snapshot" of chlorophyll distribution, due to the time taken to cover the survey area inevitably the survey represents composite of measurements spanning a period of time, typically 40 to 90 minutes. Surveys were also undertaken in differing conditions, all which may influence the results of the survey, namely:

- The level and age of cultures in existing farms (*i.e.* different filtration pressures)
- Different times (*i.e.* different states of the tide and seasons)
- Different locations with differing physical (*i.e.* flushing rates, stratification and currents) and biological (*e.g.* natural algal concentrations) regimes
- Naturally occurring levels of spatial "patchiness" within algal populations

Consequently, results from these surveys will be dependent on the conditions under which they were measured, as is undertaken in the site-specific assessments. Nevertheless, a broad-scale synthesis of these surveys is useful for identifying the approximate magnitude and extent of local area depletion by shellfish farms, and whether this is indeed a significant environmental concern. Particularly given mussel farming culture represents the scenario with the highest density of culture (animals per cubic metre) and the amount of water filtered by each animal (as highlighted by the results of Hawkins *et al.* 1999).

In order to summarise these data, a review of 36 surface chlorophyll surveys was undertaken to assess the efficacy of this method for determining the extent of depletion around existing mussel farm developments. An automated comparison of spatially-interpolated chlorophyll concentration was undertaken, where the chlorophyll concentration of the four closest farms sites within the survey area (regardless of the level of culture) was compared to concentrations outside the farm site (Table 9 and Figures 24 - 27). The results of this analysis show:



- 21 of the 36 surveys had comparatively lower concentrations within the farmed areas of between 0.96% and 14.79%.
- 4 surveys appeared to show no difference between within and outside of the farm areas.
- 8 surveys appeared to have higher concentrations within the farm areas of between 1.41% to 12.77%.
- 3 surveys either had no data or farms within the survey area.
- Table 9.Mean percentage difference of the inside farm chlorophyll concentrations relative to the outside<br/>farm chlorophyll concentrations for 18 surveys from Port Underwood and Horseshoe Bay<br/>(Pelorous Sound) and 18 surveys from Kenepuru Sound.

Location	Survey	NZMG_E	NZMG_N	Mean% Diff
				+/- SE
Port Underwood	PU01	2605557	5989004	4.2 +/- 0.1%
	PU02	2605551	5989043	-7.1 +/- 0.2%
	PU03	2605962	5989619	-4.1 +/- 0.2%
	PU04	2605249	5988259	-2.2 +/- 0.2%
	PU05	2605256	5988169	0.0 +/- 0.2%
	PU06	2605947	5989611	-4.9 +/- 0.3%
	PU07	2605931	5987470	0.0 +/- 0.6%
	PU08	2605926	5987471	0.0 +/- 0.3%
	PU09	2606616	5988193	-1.4 +/- 0.2%
	PU10	2606613	5988158	1.2 +/- 0.4%
	PU11	2605259	5984908	-1.0 +/- 0.2%
	PU12	2605248	5984896	-1.3 +/- 0.4%
	PU13	2604454	5984448	-0.9 +/- 0.4%
	PU14	2604458	5984451	5.2 +/- 0.9%
Horseshoe Bay	HB01	2589034	6019808	1.4 +/- 0.3%
	HB02	2589047	6019805	-1.6 +/- 0.4%
	HB03	2589638	6018851	2.9 +/- 0.3%
	HB04	2589627	6018820	-4.8 +/- 0.4%
Kenepuru Sound	KP01	2597860	6002591	8.4 +/- 0.5%
	KP02	2597860	6002601	-3.4 +/- 0.3%
	KP03	2599917	6002307	N/A
	KP04	2599955	6002382	N/A
	KP05	2587324	5999495	-12.3 +/- 1.4%
	KP06	2587407	5999510	-0.6 +/- 0.2%
	KP07	2588447	5999793	-6.8 +/- 0.7%
	KP08	2588481	5999808	-9.9 +/- 1.0%
	KP09	2584622	5999957	-7.8 +/- 1.0%
	KP10	2584621	5999987	3.9 +/- 1.0%
	KP11	2583662	5998404	12.8 +/- 0.6%
	KP12	2583735	5998404	-14.1 +/- 1.7%
	KP13	2593988	6002235	-8.4 +/- 0.5%
	KP14	2594528	6002333	N/A
	KP15	2594709	6003258	0.0 +/- 0.7%
	KP16	2594719	6003195	-14.8 +/- 0.6%
	KP17	2596503	6003578	-5.7 +/- 0.5%
	KP18	2596541	6003581	-4.9 +/- 0.4%

On the basis of these results alone there is some weak evidence which suggests that local depletion within mussel farming sites is occurring. However, a subjective visual assessment of interpolated survey data shows that there are possibly more sites where down-current depletion may extend outside of the farmed areas (so called "depletion shadows"). Of the 36 surveys inspected, approximately 29 (80%) of the surveys appeared to have depleted regions within or close to existing farms (Figure 24) which provides more evidence for local depletion by farms. However, a lack of control sites for these assessments it is difficult to state conclusively what fraction of the apparent depletion in the sites is a result of natural spatial variation (e.g. Figure 27) and what fraction is driven by the culture.

These results are further confounded by the fact that chlorophyll measurements are taken from a single depth. Vertical aggregations or "patches" of phytoplankton are also common and given the mixing influence of aquaculture structures (*e.g.* Stevens *et al.* 2008) it is possible that the apparent depletion shadows are partly an artefact of dilution where lower concentrations of phytoplankton are mixed into the sampling depth. Indeed, the opposite of this mechanism (higher concentrations mixing into the sample depth) may also help explain the appearance of an "enhancement shadow" seen in some of the surveys (Figure 26). Additionally, given one of the goals of the surveys to address the impacts on naturally occurring filter-feeding organisms which are likely to be situated on the seabed. Given the chlorophyll measurements relate to surface waters it is difficult to assess the relevance of these results to organisms at depth.

In order to remove any potential bias and improve the utility of these surveys, the inclusion of control sites and depth-integrated chlorophyll measurements would address the stated issues. However, these may not be practical options given the uniqueness of individual sites would make control site selection extremely subjective and depth-integrated surveys would further increase the time taken by a single vessel or survey costs of additional vessels to assist in undertaking the surveys. Despite the limitations of these surveys, if the results are assumed to be broadly representative of the water column, it appears that local depletion of the water column around mussel farms is likely and is up to about 80% of the mean ambient concentrations (Table 9) although maximum differences may be higher (approximately 50-80%). The relevance of this result to the wider ecosystem is difficult to address given that the response of competing organisms to a reduction in seston supply may not result in a corresponding linear response in growth due to base metabolism overheads. Nevertheless, it seems likely that the energy available for growth and reproduction of organisms close to aquaculture sites could be reduced by up to the 20% observed in the surveys. This would be expected to have some negative influence on the growth, fecundity and abundance of these natural populations proximate to cultured areas.

Due to the limited spatial extent of these surveys, it is difficult to assess wider area implications from these survey results. This is further complicated by the possibility of suggested far-field mechanisms, such as improved phytoplankton recovery from shifts to more bioavailable nutrients (*e.g.* Nitrate to ammonia) and increase availability of light from increases in water clarity (Broekhuizen *et al.* 2002). Consequently, the use of biophysical numerical models to assess larger area impacts will continue to prove useful.



In summary, it seems that there is evidence to suggest that local depletion around shellfish farms does occur, with a majority of the studies showing either local depletion within or close to the farm site. A maximum mean percentage decrease of about 15% is observed between the interior of the farm sites the area outside for an individual survey (KP16 - Table 9) and visual inspection of the data suggests that depletion shadows from farms may lead to localised depletions which are as high as 50% of the maximum ambient concentrations (Figure 24). More rigorous analysis of these datasets could also be undertaken; however this would not necessarily offer greater utility given the underlying systemic issues of this approach, namely a lack of control data and single depth readings. Consequently, the magnitudes seen in these surveys may serve as a guide for estimating local impacts only in a qualitative sense.















**Figure 24.** Examples of chlorophyll concentration surveys undertaken where results appear to show depletion within or close to farm sites.













**Figure 25.** Examples of chlorophyll concentration surveys undertaken around existing farm sites where results do not appear to show depletion.



**Figure 26.** Examples of chlorophyll concentration surveys undertaken around existing farm sites where results appear to show enhancement of chlorophyll.





**Figure 27.** Examples of chlorophyll concentration surveys undertaken away from existing farm sites showing natural variation in chlorophyll concentrations.



# Appendix 2. Diseases and pathology in farmed shellfish and other non-finfish species.

Over a hundred species of mussel parasites/symbionts have been reported in overseas mussel species (Webb 2007) but only the paramyxean protozoan *Marteilia* sp. and putative virally transmitted hemic neoplasia appear to be economically significant in their home waters. That these two have not been reported in New Zealand is fortunate, but it means that we cannot gauge their infectivity or virulence to Greenshell<sup>TM</sup> mussels and other native bivalves. Overseas reports in bivalves do show, however, that aquaculture-borne transmission from cultured animals to wild stocks is possible and vigilance is required to forestall such risks.

#### **Greenshell<sup>TM</sup> mussels**

Hine (1989) reported no disease-associated mortalities in Greenshell<sup>TM</sup> mussels. Similarly, Hine (1996) included no listed serious or potentially serious pathogens in New Zealand blue mussels *Mytilus* spp. or Greenshells<sup>TM</sup>. A recent review on mytilids with particular emphasis on *P. canaliculus* (Webb 2007) indicates that there are no particularly destructive diseases of mussel species in New Zealand, with the exception of a digestive viral disease. That disease was first noted by Jones *et al.* (1996) who reported mortalities in cultured Greenshell<sup>TM</sup> mussels in the outer Marlborough Sounds of 50-100% associated with virus-like particles and digestive tubule damage. They also reported similar infections in mussels from Westport. Subsequent surveys (Hine 2002a; S Webb, pers. obs.) have shown that although present, the disease appears to have receded in importance to sporadic events and low infection levels. The condition also affects scallops and clams in New Zealand and other molluscs elsewhere. Besides being found in other New Zealand molluscs, viruses producing similar digestive tissue effects have been reported in Australia, Scotland, Denmark, and elsewhere (Bower 2001).

Subsequently, Hine (2002a) reported that Greenshell<sup>TM</sup> mussels examined from the Marlborough Sounds and Coromandel appeared to be in good health overall despite recording low levels of apparent digestive viral infection. Jeffs *et al.* (1999) echoes these reports when he cites mortality problems associated with an unenveloped RNA virus as of greatest concern in Greenshell<sup>TM</sup> mussels and that, of several other parasites found, none caused significant mortalities. In addition to the digestive viral disease, a suite of other parasites was also mentioned by Hine (1997) however none was deemed a serious cause of mortality.

Another pathogen that warrants mention is the parasite APX, which is reported from New Zealand only (Diggles *et al.* 2002; Hine 2002b) and has been found in mussels from the Marlborough Sounds and also occurs commonly in dredge oysters *O. chilensis* (also known as flat oysters) from all around the coast (Diggles *et al.* 2002; Hine 2002b). In dredge oysters APX can cause a significant condition referred to coccidiosis (Hine & Jones 1994); however, its effect on mussels is less noteworthy.

In summary, cultured Greenshell<sup>TM</sup> mussels appear to present no major threat to wild molluscs, as wild Greenshell<sup>TM</sup> stocks appear to harbour all pathogens with the exception of APX. Since



APX is also found in dredge oysters, however, there would remain a reservoir of infection even in the absence of Greenshell<sup>TM</sup> mussel culture.

#### Potential disease risks

Exotic pathogen threats to Greenshell<sup>TM</sup> mussels can only be speculated upon. In this category, *Marteilia* spp. and disseminated haemic neoplasia (a molluscan leukemia) were determined by Webb (2007) as the most likely overseas threats. The potential consequences of introduction of hosts with these conditions and possible pathways by which more susceptible hosts could emerge are discussed below.

The greatest potential threat to New Zealand Greenshell<sup>TM</sup> mussel aquaculture appears to be posed by parasites introduced by invading species of blue mussel (*e.g. Mytilus edulis*). These common ship-borne fouling organisms are a likely source of overseas pathogens. Hybridisation of invasive with indigenous blue mussels (*M. galloprovincialis*) presents a further potential pathology hazard by production of a more susceptible reservoir host suitable for these pathogens. Evidence for such a risk is found in Beaumont *et al.* (2004) who report depressed performance of *M. edulis* x *M. galloprovincialis* hybrids when compared with pure species. In addition, Fuentes *et al.* (2002) report the lower viability of hybrids challenged by heat shock or infection with *Marteilia refringens*.

The physical coincidence of hybridising mussels and pathogens in New Zealand waters is possible: *Mytilus* species are likely to be arriving regularly in New Zealand and they could be carrying pathogens such as *M. refringens or M. maurini*. Some protection might be afforded by the currently remote known range of these *Marteilia* spp. [western Europe and the Mediterranean (Bower 2007)] but this restricted range may be a more a function of survey effort rather than actual distribution. Arrival of *M. edulis* facilitates hybridisation of *M. edulis* x *M. galloprovincialis* and the resulting hybrid, because of its increased susceptibility to *M. refringens*, might provide a more accommodating reservoir host with a consequent increase in numbers of pathogen transmission stages. Hypothetically, if *Mytilus* spp. were to become infected with *Marteilia* and if appropriate intermediate hosts (or local substitutes) were present, then there is the possibility of transmission to Greenshell<sup>TM</sup> mussels. We note, however, that the pathological threat to Greenshell<sup>TM</sup> mussels posed by *Marteilia* from Greenshell<sup>TM</sup> mussels is because of resistance by the mussels, lack of infection opportunity or perhaps because of insufficient geographic range of sampling specifically for *Marteilia*.

Hemic neoplasia may benefit from similar conditions. It is reported in *Mytilus* species and is associated with high mortalities. Hybrids of *M. edulis* and *M. galloprovincialis* (Fuentes *et al.* 2002) have been reported with elevated prevalences as compared with pure species, and invasion dynamics are likely to produce hybrids contemporaneously with the arrival of the neoplasia. The potential thus exists for production of more infected susceptible hosts and greater water load of transmission stages. It is known that this condition can be transmitted by cohabitation (Bower 2006) thus suggesting a direct life cycle which eliminates the need for a local intermediate host and simplifies transmission. Currently, we cannot say how harmful or

otherwise these two pathologies might be to Greenshell<sup>TM</sup> mussels. Our only guide is that they are both damaging to *Mytilus* populations.

Overseas, the aquabirnavirus Infectious Pancreatic Necrosis virus (IPNV) has also been detected in *M. edulis* (VPS 2000). It is a common virus of salmonids and is also a suspected clam pathogen in Taiwan. Similarly, Kitamura *et al.* (2007) report finding an aquatic birnavirus (ABV) in *M. galloprovincialis* where the mussel was acting as a reservoir host for infections in the Japanese flounder *Paralichthys olivaceous*. This internationally significant disease of world wide distribution is reported in healthy King salmon (*Oncorhynchus tshawytscha*) returning from the sea on the east coast of South Island, New Zealand (Diggles *et al.* 2002). Although not detected in New Zealand mussels, the possibility of *P. canaliculus* harbouring this virus, at least temporarily, finds support in the reports by Lewis *et al.* (1986) and Greening *et al.* (2001) where polioviruses and enteroviruses have been shown to persist in *P. canaliculus* after experimental exposure. Caution is clearly required in polyculture, as mytilids might harbour viruses with consequent threat to susceptible fish.

## **Pacific oysters**

*Crassostrea gigas* was first reported in New Zealand (Dinamani 1971) from Northland; it subsequently appeared (Jenkins & Meredyth-Young 1979) in the South Island and since then the parasites of this oyster in New Zealand have received significant attention. There are no OIE listed (OIE 2001) serious parasites/pathogens of *C. gigas* in New Zealand (Diggles *et al.* 2002). Nevertheless several diseases and parasites associated with New Zealand Pacific oysters have been reported, most of which are also globally ubiquitous and pose some commercial threat to oyster production (especially in hatcheries). These include vibriosis (Overseas incidences are documented in Bower 2002), rickettsiosis, and planocerid flatworms (Diggles *et al.* 2002); spionid mud-worms (Handley 1995; Handley & Bergquist 1997) and ostreid herpes virus OsHV-1 (Hine *et al.* 1992), which infects oyster larvae and spat. In the latter case, however, a recent survey (Webb *et al.* 2007) has not detected this virus in *C. gigas, Ostrea chilensis* or a range of other New Zealand adult bivalves. Organisms of insignificant impact (Dinamani 1986) include turbellarians, chironomids, nematodes, mudworms and pea crabs. Hine & Jones (1994) mentioned the copepod *Pseudomyicola spinosus* but asserted that it appears to have little effect.

Despite the above occurrences, New Zealand oysters are generally healthy: a survey of 290 Northland oysters failed to detect any significant parasites (Hine 1997). In addition, "with the exception of *Perkinsus olseni* only relatively trivial infections occur in New Zealand commercial bivalves. "...Pacific oysters in particular consistently appeared to be in good health" says Hine (2002a). In the light of the currently known local diseases, it can be inferred that culture of *C. gigas* in New Zealand is unlikely to pose a threat to naturalised *Crassostrea* or other species. Conversely, so far in New Zealand, *C. gigas* has not suffered significant or unexpected effects from indigenous pathogens, which suggests minimal threat from that quarter.



#### Pathogens in overseas populations of Pacific oysters

- The oyster parasite *Bonamia ostreae* was introduced into France from California on seed oysters of *O. edulis* (Carnegie 2005). It has since spread to other European countries and was also introduced to Washington from California (Straus *et al.* 2008).
- The paramyxean protozoan *Marteilia sydneyi* is thought to have extended its range by aquaculture of *Saccostrea glomerata* transplantation (Carnegie 2005; Bower & Kleeman 2007).
- The protozoan parasite *Haplosporidium nelsoni* introduced in the Pacific oyster *C. gigas* is implicated in mortalities of native Chesapeake oysters (Torchin & Kuris 2005).
- The bacterial disease nocardiosis, associated with Pacific oysters, originated in Japan and appears to have spread to California, Washington, and British Columbia (Straus *et al.* 2008).
- The withering syndrome organism, *Xenohaliotis californiensis*, was probably spread in California by outplanting of infected hatchery reared abalone (Friedman & Finley 2003).

Pacific oysters appear to evade many of the disease issues that beset other oysters (Elston 1993; FAO 2006). For instance *C. gigas* is partially resistant to infection and disease caused by *Perkinsus marinus* (Bower 2006a) which affects *Crassostrea virginica* severely. Bower (2007b) report that, in experimental challenges, *C. gigas* seems to be more resistant to *Mikrocytos mackini* than some other oysters.

Despite this, a number of significant pathologies do affect C. gigas. Summer mortalities of oyster seed have been linked to herpes virus in California but a causal association has not been confirmed (Friedman et al. 2005). Recent unpublished work from France has produced more convincing circumstantial evidence linking herpes virus with such mortalities. Oyster velar virus disease, caused by an irido-like virus, can result in near 100% mortality in affected hatchery tanks (Bower 2001d). Hinge ligament disease caused by bacteria of *Ctyophaga* spp. can affect juvenile oysters (~1cm) especially in warmer water (Bower 2001e). Another pathogenic bacterium Nocardia crassostreae can occur in C. gigas and O. edulis cultivated nearby (Bower 2006b). Nocardiosis, associated with Pacific oysters, originated in Japan and appears to have spread to California, Washington, and British Columbia (Straus et al. 2008). Rickettsia-like and Chlamydia-like organisms (Rickettsiales) are intracellular parasites (Bower 2006c) in which heavy infections in C. gigas are reported to have caused gill lesions and mortalities. Ciliate infections have been associated with mortalities exceeding 50% in oyster seed (Bower 2001c) and similar ciliates have been seen in New Zealand oyster spat (S Webb, pers. obs.). Denman Island Disease Mikrocytos mackini (OIE listed) prospers due to either its increased pathogenicity or the increased susceptibility of Pacific oysters in areas where there may be several months at water temperatures below  $10^{\circ}$ C (Bower 2007b). The protozoan Haplosporidium nelsoni (OIE listed) introduced in the Pacific oyster C. gigas is implicated in mortalities of native Chesapeake oysters (Torchin & Kuris 2005). Although introduced in C. gigas, H. nelsoni appears to have greater pathological impact on C. virginica (Bower 2007c). Marteilioides chungmuensis can infect C. gigas and impact on marketability by degrading appearance (Bower et al. 2006). M. refringens (OIE listed) is thought to have occurred in C. gigas, but has been confirmed in O. edulis, M. edulis, Cardium edule, C. virginica and Ostrea



*chilensis* (Bower 2007a). The fungus *Ostracoblabe implexa* occurs in *O. edulis*, *C. gigas*, *Saccostrea cucullata* and less severely *Crassostrea angulata*. It grows through the shell to the inner surface and can elicit production of brown wart-like nodules - water temperatures exceeding 22 °C for more than two weeks favour its development (Bower 2001a). *Echinocephalus crassostreai* (Nematoda) has insignificant effect on the oyster, but it can cause of human health problems (Bower 2001b). The customary eating of oysters raw facilitates human infection.

Previous oyster introductions or translocations of native species are reported (Ruesink *et al.* 2005; Carnegie 2005) to be a significant cause of molluscan disease outbreaks. Clearly, a number of diseases mentioned would have significant impact on the New Zealand oyster industry if they were to become established here. In some cases they may also impact on the local molluscan fauna. Good biosecurity practises and surveillance are probably the best way of managing the threat. Specific mitigation factors should be devised from best practice elsewhere should one of these exotic diseases become established.

#### **Ecological factors**

Despite being a recently introduced species, New Zealand *C. gigas* have been reported to show no evidence of reduced genetic variation (Smith *et al.* 1986). This lack of a founder effect is surprising, but on the positive side it would suggest that the naturally high resistance of this species to many infections has not been degraded by genetic bottlenecks in New Zealand populations. Of course, further work is required to confirm this. Such work would also be an opportunity to ascertain gene flow between naturalised and cultivated New Zealand *C. gigas*. This might also afford insights into potential pathogen flows between the two populations.

Introduced organisms, even if they do bring parasites with them, could benefit from dislocation of the parasite life cycle. For instance, some parasites such as *Marteilia* spp. and *Haplosporidium* spp. appear to have an indirect life cycle requiring intermediate hosts. These hosts may have been left behind and there may be no local substitutes – we currently cannot make informed inferences on this. It is possible that potential intermediate hosts (if present) could be part of the suite of fouling organisms: this would afford opportunities both in progressing life cycle studies and as possible control measures.

The apparent advantage of *C. gigas* in being more resistant yet not immune has a darker side. It might also serve as an asymptomatic reservoir of pathogens damaging to other more susceptible species - the account above of diseases carried by *C. gigas* clearly illustrates that a range of other molluscs can be hosts. Although New Zealand may lack the exact species mentioned, congenerics and others of close affiliations do occur in New Zealand waters and could be similarly affected. It follows that should New Zealand *C. gigas* suffer an incursion by an exotic disease, it is likely that not only will naturalised *C. gigas* suffer, but so too would other molluscan species. Despite these pessimistic possibilities, the effect of invasive molluscs can be unpredictable. An example is provided by Thieltges *et al.* (2008) who report on the mitigation of the parasite burden of *M. edulis* by the presence of introduced Pacific oysters (*C.*  *gigas*) and American slipper limpets (*Crepidula fornicata*). It appears that the introduced species diverted the trematodes from their usual hosts, thus reducing infection levels.

The above examples, both optimistic and pessimistic, highlight the range of possible outcomes. Obviously, further studies are required before we can begin to understand the factors that govern these. Until the current paucity of information is remedied any projection must be largely speculative – in which case a precautionary approach should be adopted that assumes a worst case until proven otherwise. Further work will allow us to relax these strictures.

## **Flat oysters**

The most significant pathogen found in New Zealand *O. chilensis* is the OIE (2000) listed haplosporidian protozoan *Bonamia exitiosa* (Diggles *et al.* 2002) which can cause significant to severe mortalities in adult oysters. Second in importance is apicomplexan X (APX); this protozoan is reported from New Zealand only, where it occurs in mussels from the Marlborough Sounds and in flat oysters from all around the coast (Diggles *et al.* 2002). Its presence is thought to predispose the oysters to *B. exitiosa* infection (Hine 2002). A similar coccidian has been reported from the kidneys at low prevalences (Hine & Jones 1994). A potential problem in hatcheries is the ostreid herpes virus OsHV-1. This virus, of worldwide distribution, has been reported in New Zealand *O. chilensis* (Hine *et al.* 1998; Diggles *et al.* 2002) and has been associated with significant larval mortalities. A more recent survey (Webb *et al.* 2007) has not detected this virus in *O. chilensis* or a range of other New Zealand adult bivalves.

Parasites and pathogens of secondary importance to *O. chilensis* include the following. Mudworm infestations by the spionids *Polydora* spp. and *Boccardia* spp. can cause embrittlement of the shell and stimulate production of thin nacre-covered mud blisters on the inner shell surface (Diggles *et al.* 2002). This problem occurs as a nuisance to many bivalve species and can also affect paua. *Microsporidium rapuae*, a protozoan, (Hine & Jones 1994; Webb, pers obs.) appears to have no pathological effect. Sporocycts of the digenean trematode *Bucephalus longicornutus* (Jones 1975, Hine & Jones 1994; Webb, pers obs.) are commonly encountered, usually at low prevalences. Although individual oysters are often heavily parasitised the pathogen is at worst a nuisance. In common with its occurrence in other bivalves, the copepod *Pseudomyicola spinosus* (Jones 1975) has minor or insignificant impact on the host. Rickettsia-like organisms (RLOs), although they have been implicated in mass mortalities in scallops, generally produce no gross signs of pathology (Diggles *et al.* 2002) in *O. chilensis*. Neoplasms, such as hemic neoplasia, germinomas and seminomas are reported by Hine (1997) to occur at prevalences of below 1%.

# Scallops

Most scallops surveyed for digestive epithelial virosis (DEV) have high prevalence and high relative intensity (Hopkins *et al.* 2003; Hopkins & Webb 2004; Webb & Hopkins 2005; Webb 2006; Webb & Govier 2006; Webb & Govier 2007). Previous reports on this virus include: Jones *et al.* (1996); Hine & Wesney (1997); Diggles *et al.* (2002). The latter authors assert that all scallops examined from around New Zealand have these viruses and that infections may possibly become pathogenic if they reach high levels. Rickettsia-like organisms (RLO)



infections are ubiquitous and severe infections have been implicated in periodic mass mortalities (Diggles *et al.* 2002). Surveys over several years have confirmed the high prevalence and intensities of infections (Hopkins *et al.* 2003; Hopkins & Webb 2004; Webb & Hopkins 2005; Webb 2006; Webb & Govier 2006; Webb & Govier 2007). Surprisingly, other than the gill involvement, there are few signs of pathology. RLOs also occur in kidney, muscle and digestive epithelial tissue at low prevalences and intensities with no apparent health impact (Webb & Govier 2007). Unidentified inclusions were recently found in a survey of Marlborough Sound scallops (Webb & Duncan 2008). The condition consists of many 10-15  $\mu$ m inclusions at moderate to high prevalences and intensities in the mantle, palps, digestive gland, kidney and gonad tissues. Despite high prevalence and intensity, further work is needed to gauge pathogenicity of this condition.

Other parasites are of minor significance. Prokaryotic mycoplasmas are reported from the blood cells (Diggles et al. 2002). The turbellarian Paravortex occurs (Woods & Hayden 1998) in apparently healthy scallops. Hopkins & Webb (2004) discuss the pathological significance of *Paravortex*. No pathological effects (S Webb, pers obs.) have been noted in association with any infections. Spionids - annelid worms responsible for mud blisters - occur at low prevalences; no pathological effects were associated with any of these occurrences (Hopkins et al. 2003; Hopkins & Webb 2004; Webb & Hopkins 2005; Webb & Govier 2006; Webb & Govier 2007). The copepods *Pseudomyicola* sp. and *Lichomolgus* sp. appear to be innocuous in scallops even at intensities comparable to those in *Mytilus* spp. where reduction in condition has been noted (Caceres-Martinez et al. 1996). Webb & Govier (2006) discuss the minor pathological threat posed by these copepods. Nematodes (roundworms) occur at low prevalences (usually below 5%) and intensities with no signs of associated tissue changes or damage. They are likely to be fortuitous inclusions rather than dedicated parasites (Webb & Hopkins 2005). Crustacean ostracods are uncommon, and probably commensal or accidental rather than pathogenic (Webb & Govier 2006). The pea crab *Pinnotheres* sp. has been seen at prevalences below 5% (Hopkins et al. 2003; Webb & Hopkins 2005). Its pathological effect is negligible. Hopkins et al. (2003) review Pinnotheres and its relationship with bivalve hosts. A Nematopsis-like gregarine (Protozoa: Apicomplexa) at low prevalences and intensities has been noted in the mantles of scallops (Webb & Govier 2007; Webb & Duncan 2008). Similar parasites are common in many bivalve hosts and are considered to be of minor pathological significance even in heavy infections (Jones 1975). One example of a larval tetrarhynchidean or lecanicephalidean tapeworm was found encysted in connective tissue under the digestive epithelium of a scallop from Tasman Bay (Webb & Govier 2007). These cestodes usually develop to adults in sharks and rays and thus pose no threat to human health. Similar occurrences have been reported elsewhere (Getchell 1991). Dark structureless granules were found in the nephridia (kidneys) at prevalences from below 10% to over 80%. There were no other signs of pathology or pathogens in the affected scallops. Benninger & Pennec (1991) ascribe the granules to phosphate metabolism or for the dexotification of heavy metals. One scallop from Golden Bay contained a metacercarial (Trematoda: Digenea) cyst in the mantle tissues (Hopkins & Webb 2004). Such metacercariae present a pathological threat only when there are many (dozens, hundreds or thousands) present in the host. This occurrence is pathologically insignificant.



#### Blue mussel (Mytilus galloprovincialis)

No mussels in New Zealand have been reported with any pathogens on the Office International des Epizooties (OIE) list of important diseases (Webb 2007). Moreover, few *M. galloprovincialis* parasites of any kind are reported from New Zealand. Jones (1975) and Hine (1997) mention the digenean Tergestia agnostomi, the copepod Pseudomyicola spinosus and the pea crab Pinnotheres sp., of which Hine (1997) asserts that none are apparently pathogenic. In a further work Jones *et al.* (1996) report infections with a digestive epithelial virosis. This paucity of listed Mytilus parasites in New Zealand probably reflects its lesser commercial importance rather than any biological propensity. It is likely that the parasite fauna of the blue mussel is comparable to that of the green mussel, but with the data currently available, pathogen resistance differences between New Zealand Mytilus and Perna cannot be determined.

The greatest potential disease threat posed by local *M. galloprovincialis* is their possible facilitation of establishment by serious exotic diseases such as Marteilia refringens or M. maurini and hemic neoplasia. Invading species of blue mussel (*e.g. M. edulis*) are common ship-borne fouling organisms that can hybridise with indigenous blue mussels (*M. galloprovincialis*) to produce a susceptible reservoir host suitable for these pathogens. Invading *Mytilus* species could also be carrying the pathogens. See Webb (2007) for details of other significant exotic *Mytilus* pathogens.

Overseas, the aquabirnavirus Infectious Pancreatic Necrosis virus (IPNV) has been detected in *M. edulis* (VPS 2000). It is a common virus of salmonids and is also a suspected clam pathogen in Taiwan. Similarly, Kitamura *et al.* (2007) report finding an aquatic birnavirus (ABV) in *M. galloprovincialis* where the mussel was acting as a reservoir host for infections in the Japanese flounder *Paralichthys olivaceous*. IPNV is an internationally significant disease of worldwide distribution is reported in healthy *O.s tshawytscha* returning from the sea on the east coast of South Island, New Zealand (Diggles *et al.* 2002). Caution is clearly required in polyculture, as mytilids might harbour such viruses with consequent threat to susceptible fish.

# Paua (Haliotis iris)

A range of disorders have been noted in *H. iris*. Diggles & Oliver (2005) report haplosporidia, epithelial erosion, rickettsial inclusions in gut, protozoa in foot epithelium, bacterial infection (see also Bower 2006a), non-specific necrosis, granuloma-like lesions, haemocytic neoplasialike inflammation and gregarines (apicomplexans). Diggles *et al.* (2002) report pustule disease caused *Vibrio* bacteria. Paua also exhibit a fungal shell mycosis (Grindley *et al.* 1998) as well the shell boring *Spionid* mud worms *Polydora* and *Boccardia* (Diggles *et al.* 2002; Bower 2006e) that can be a problem in culture. Severe cases of mudworm can cause significant shell embrittlement (S Webb, pers. obs.). In addition to shell damage, there can be loss of condition: *H. iris* infected with *Polydora hoplura* can be underweight and produce abnormal deposits of conchiolin (Diggles & Oliver 2005). Despite the significant nuisance of some, none presents an insurmountable obstacle to the New Zealand abalone industry. Hine (1997) in his review of health in commercially important New Zealand molluscs mentions only fungal shell disease and even that he says is not a cause of significant mortality. Diggles & Oliver (2005) add to this that the haplosporidian has been associated with mortalities. Potential problems could arise from rickettsia, granuloma-like lesions, inflammatory lesions suggestive of hemocytic neoplasia, mudworm and fungal infections as discussed by Diggles & Oliver (2005). More needs to be done on investigating the husbandry factors influencing these agents.

Since some of these pathogens are apparently found only in New Zealand waters, it is difficult to gauge their potential threat to foreign abalone. In cases such as that of our fungal mycosis (Bower 2006f) there are no extant control methods and prevention is the only option. Rigorous examination and quarantine regimes before introduction to new foreign habitats is essential.

Although apparently uninfected, abalone in New Zealand are potentially at risk from an indigenous parasite, *Perkinsus olseni*, which currently is reported in the Northland bivalves *Austrovenus stutchburyi*, *Macomona liliana*, *Barbatia novaezelandia* and *Paphies australis* (Diggles *et al.* 2002). It is curious that in the higher water temperatures of Australia the, apparently, same *Perkinsus* species can infect *H. rubra*, *H. laevigata*, *H. cyclobates* and H. *scalaris* (Bower 2007). The lack of host specificity shown by this parasite suggests that paua would not be immune in favourable circumstances. Perhaps climate change could be the factor that allows this threat to materialise.

#### Exotic abalone pathogens

Reported exotic abalone pathogens that could impact on New Zealand abalone include, amyotropia (probably viral), withering syndrome from the West Coast of United States of America (Diggles *et al.* 2002; Bower 2006d); the shell dwelling sabellid *Terebrasabella heterouncinata* (Bower 2006b); *Labyrinthuloides haliotidis* a protist in *H. kamtschatkana* and *H. rufescens* (Bower & Meyer 2005) and the kidney coccidia *Margolisiella* (=*Pseudoklossia*) *haliotis* from the West Coast of United States of America (Bower 2006c). A more distasteful but possibly less likely threat comes from the nematode *Echinocephalus pseudouncinatus* (Bower 2001). It weakens the foot muscle and allows easier detachment from the substratum. The usual final hosts are certain sharks and rays, but human consumption of the live worms in undercooked abalone may allow migration of the larvae through human tissues.

The most immediate exotic threat to New Zealand abalone is from viral ganglioneuritis which now has been reported from farmed Australian *H. laevigata* and *H. rubra* (Hooper *et al.* 2007). It is thought to have come from the Far East where farmed abalone have been reported with similar herpes-like viruses (Wang *et al.* 2004; Chang *et al.* 2005). Mortality attributable to this virus has occurred on some farms (Hooper *et al.* 2007) and there is evidence to suggest that the virus has spread to wild populations causing significant mortality events among abalone, and possibly other gastropods (Hine 2006). As to the hazard facing New Zealand, Hine (2006) concluded that the taxonomy and geographic isolation of *H. iris* in New Zealand is such that it is likely to be currently free of the virus. Clearly, this virus could have significant impact on New Zealand paua if it became established here.


# Appendix 3. Consequences of the movement, mixing and interbreeding of genetically distinct stocks of Greenshell<sup>™</sup> mussels in New Zealand

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First draft - 24 November 2008

## Rationale

This section deals specifically with the genetic and associated fitness consequences of the movement of Greenshell<sup>TM</sup> mussels around New Zealand, whether this be via transfer of spat from Kaitaia to aquaculture sites, or via other means such as hatchery-based breeding of mussels for use (transplant out to) aquaculture sites.

### Background

The Greenshell<sup>TM</sup> mussel, *Perna canaliculus* Gmelin 1791, is endemic to New Zealand and is distributed from the far north (approx  $34.5^{\circ}$  S) to as far south as Stewart Island (approx  $47^{\circ}$  S). It is not found on offshore islands such as the Chatham Islands or any of the Subantarctic Islands. The mussel is the focus of a significant aquaculture industry, which had an estimated export value in 2007 of \$174 million to the New Zealand economy (http://www.seafood.co.nz/greenshell). As outlined earlier on in this report, approximately 80% of the spat (small mussels) used by industry are collected on drift *Sargussum* seaweed on Ninety Mile Beach in the far north (approx  $35^{\circ}$  S) which is then trucked to three main aquaculture centres – the east coast of the Coromandel Peninsula (approx  $37^{\circ}$  S), the Marlborough Sounds (approx  $41^{\circ}$  S), and Stewart Island (approx  $47^{\circ}$  S) (Hickman 1983). The purpose of this section is to review the possible genetic and fitness consequences of such mass movement of mussels around New Zealand.

## Introduction

Evolutionary theory suggests that populations which experience different environmental regimes (temperature, salinity, food availability, sedimentation, wave exposure *etc.*) as well as differing biological regimes (*e.g.* predation, parasitism, disease *etc.*) will become genetically adapted to the local conditions via the process of natural selection. Because natural selection acts at the level of the individual and not at the level of the population (at least in this particular case), this localised adaptation allows individual mussels to maximise their fitness (their reproductive contribution to the next generation) throughout their life spans. For the purposes of this argument, this adaptation can be thought of as occurring at small spatial scales, of the order of hundreds to a few thousand metres (*e.g.* all the mussels within one bay or on one stretch of rocky coastline). Neighbouring populations (*e.g.* 

those in an adjacent bay) may be adapted to similar environmental conditions because the physicochemical properties of the two bays are similar or almost identical. Alternatively, the mussels may be differently adapted because the two bays are very different in their environmental properties. For example, one bay may have riparian input which results in fluctuating salinity levels and also in variable particulate food supply (quantity and quality of particulates) over time, whereas the adjacent bay may not have any riparian input and may be less variable in terms of its physico-chemical properties over time. At a larger spatial scale, of the order of hundreds of kilometres, it is expected that New Zealand mussels will be genetically adapted to different environments because the country (the three main islands) spans 13° of latitude, from the warm subtropical north to the cold temperate south. Natural selection is therefore thought to give rise to what are known as "co-adapted gene complexes". These can be thought of as an assortment of genes which function very efficiently together (*i.e.* they confer high fitness on the individual) in the given environment. As outlined above, different environments require different genes to confer high fitness – this is the main (but not only) reason why genetic variation is thought to exist among populations.

The difficulty from a technology perspective with the above is that we are only now starting to identify the genes which may be responsible for fitness differences among individuals, whether these are the genes that respond to environmental variation (*e.g.* heat shock proteins) or those that regulate processes such as growth, reproduction and development. Until very recently this has meant that it has not been technically possible in almost all cases to quantify genetic differences among populations based on the natural selection response of the individuals within that population (historically, we have used surrogate genetic markers to quantify similarities or differences among populations and individuals). Recent technological advances in rapid whole genome sequencing and the ability to process huge amounts of information (*e.g.* as single nucleotide polymorphisms – SNPs) are now starting to open the door into the realm of a new approach to quantifying and understanding genetic variation, how this is influenced by environmental variability (*e.g.* via natural selection), and how this contributes to fitness differences among individuals.

The view outlined in the first paragraph of this section reflects the sessile adult life stage, but it must be remembered that mussels reproduce by spawning gametes directly into the sea, where fertilisation is external. Mussels are highly fecund. One large female can produce tens of millions of eggs, with the result that the reproductive output of a single spawning event may give rise to billions of offspring. The planktotrophic mussel larvae (a feeding stage) spend 3-4 weeks in the water column (Hayden 1994), while pediveligers remain planktonic for several weeks if suitable substrate for settlement is not encountered (Buchanan 1994). Although these larvae do have some limited swimming capacity, they are viewed as being passive particles (propagules) in the water column. As such, they are moved around by wind and tide-driven surface water circulation until they are developed enough to settle at a site, where they will undergo metamorphosis and achieve the sedentary adult life style. Thus, adults which are presumed to be adapted to their local environment give rise to offspring (larvae) which are likely to also be adapted to that environment, but which spend sufficient time being moved around in the water column that we can reasonably assume that most newly settling spat at any site are in fact immigrants and did not originate from the population into which they recruit (e.g. Tracey et al. 1975). While there are certainly well documented cases of self-recruitment (reviewed by Swearer et al. 2002) and there is increasing evidence that this phenomenon is not as rare as first thought (e.g. Wood &

Gardner 2007), in the context of the extent and patterns of larval connectivity ( = gene flow) among *P*. *canaliculus* populations, it's safe to say that self-recruitment is not an important consideration in the present situation. Because of the vast numbers of offspring produced by broadcast spawning species such as *P. canaliculus*, and because of the highly dispersive nature of the larval stage (an evolutionary adaptation to promote gene flow among populations and the colonisation of new environments), it has been reasonably assumed that connectivity among populations is high.

The final point for consideration is the view that the sea (including coastal regions) is large and open and fully connected, without barriers or impediments to the movement of biota. Historically, this idea arose out of the obvious vastness of the sea and the view that all marine regions are connected by seawater. From a marine perspective, it is the land that forms islands which are separated by the sea, and it is the sea that provides a continuous environment, even if the physical, chemical and geomorphological properties of the sea (and its substratum) vary in space and time. This view of the sea as continuous and without boundaries was at least in part reinforced by many population genetic (allozyme) studies from the 1970s onwards which often demonstrated an apparent absence of genetic differentiation, or at least very low levels of genetic differentiation, among populations which were hundreds or even thousands of kilometres apart (e.g. Scheltema & Williams 1983; Hunt & Ayre 1989; Creasey et al. 1996; Murray-Jones & Ayre 1997; Apte & Gardner 2001). This apparent lack of genetic differentiation was often explained in terms of high levels of gene flow ( = high levels of genetic mixing among populations) which in turn resulted in genetic homogeneity among populations (note that theory suggests that one recruit per generation is all that is required to be shared between populations to prevent genetic divergence between those populations). This was viewed as only being achievable because of the connectivity of many or all marine environments by the medium of the sea itself. More recent studies, on both hydrodynamic processes and on genetic connectivity, have shown that this is not the case. Many coastal regions are characterised by distributional discontinuities or barriers to gene flow despite the continuous nature of seawater. Such barriers (which may be variable in time or space) may include features such as upwelling and downwelling (e.g. Dahlhoff & Menge 1996; Apte & Gardner 2002), riparian flow with its associated freshwater and sedimentary load (e.g. Koehn et al. 1980; Gardner & Palmer 1998), gyres which entrain and trap propagules (e.g. Chiswell 2000), or may include geomorphological forms such as headlands or long stretches of habitat that are unsuitable to the organism in question. All of these features are now known to contribute to a breakdown of the "continuous nature" of the sea (in particular in coastal regions), with the result that genetic heterogeneity among populations is now known to be much more prevalent than was thought as recently as ten years ago.

# Genetic variation and genetic connectivity among populations of P. canaliculus

We can only start to assess the fitness consequences of the interbreeding of distinct genetic stocks if such stocks exist and if we know where they occur. Genetic variation, and components of it such as gene flow, can be assessed using a variety of different approaches: each has its own strengths and weaknesses. Interpretation of the data must therefore be understood in this context because different markers are informative at different levels and in different (not necessarily concordant) ways.

The earliest studies of genetic variation involved the use of allozymes, which are biochemical (protein) markers. Traditionally, allozymes are viewed as being selectively neutral (that is, they are



not under the influence of selection or if they are, that selection is so slight as be negligible). Neutral markers have the benefit of not reflecting small-scale or large-scale spatial or temporal variability which may result as a consequence of natural selection (*e.g.* environmental variability). As such, allozymes can be good tools for estimating gene flow among populations, but they are/were thought to not reflect genetic differences which may exist as a consequence of natural selection. A notable exception to this is work at the LAP locus (one of many different aminopeptidase loci) in blue mussels of the genus *Mytilus*. This locus controls internal cell volume and concentration (*i.e.* it is associated with osmoregulation) and because of this ecophysiological role it is possible to observe profound differences in individual gene frequencies as a consequence of LAP genotype-dependent selection for/against individuals in low (fluctuating) versus high (constant) salinity environments (*e.g.* Koehn *et al.* 1976, 1980; Hilbish *et al.* 1982; Hilbish & Koehn 1985; Gardner & Kathiravetpillai 1997; Gardner & Palmer 1998).

The first allozyme surveys of New Zealand Greenshell<sup>TM</sup> mussels reported different findings and interpreted these to imply very different types of population genetic structuring across the country. Smith (1988) reported significant heterogeneity between two northern and four southern populations, which led him to suggest that local hydrography, as well as genetic-physiological adaptation to different thermal environments, might partially isolate mussel populations, which could result in a warm water-adapted northern group and a cold water-adapted southern group, between which there only was limited gene flow. Gardner et al. (1996a) found no evidence of a north-south genetic split and explained the population genetic structuring that they observed by an isolation by distance model (populations which are geographically close share high levels of genetic similarity, populations which are far apart share low levels of similarity). Gardner et al. (1996b) compared allozyme variation between the wild (naturally occurring) mussels from Wellington Harbour (North Island) and mussels from the cultured population at Beatrix Bay, Marlborough Sounds (South Island). Both sites are at approximately the same latitude (~41.5°S). Mussels from the two sites exhibited different patterns of genetic linkage (non-random genotypic frequencies), indicating that the genotypic disequilibrium (the pattern of association between alleles at different loci) was different between the two populations. This can arise as a result of different selection pressures at the two sampling locations which favour different combinations of genotypes. Thus, different non-random genotypic associations are expected to originate and be maintained at different geographic locations because of the differing selection pressures (physical, chemical, and biological) that characterise the different sites. This is the concept of the co-adapted gene complex (see above). Most recently, and in the largest survey to date in terms of both numbers of populations and individuals, Apte & Gardner (2001) examined allozyme variation and found that a model of panmixia (wide spread gene flow resulting in no significant genetic differences among populations) best explained the observed genetic variation among the 35 assayed populations. Thus, while different studies have identified different models of genetic structuring within populations of *P. canaliculus* (a putative north-south split; an isolation-by-distance model; and a panmixia model), all studies have demonstrated high levels of genetic variation within the species, the exact importance of which remains unknown.

The most recent studies of population genetic variation in *P. canaliculus* have employed modern molecular approaches. These studies have examined genetic variation using markers such single-stranded conformational polymorphisms (SSCPs - Apte & Gardner 2002), randomly amplified

polymorphisms (RAPDs – Star *et al.* 2003) and microsatellites (Wei *et al.* 2009) in both the mitochondrial and nuclear genomes (these are different and are physically unlinked) of Greenshell<sup>TM</sup> mussels. There are two significant findings arising from this research.

- A pronounced genetic discontinuity exists among Greenshell<sup>TM</sup> mussel populations at ~42°S, (1)such that a northern group can be clearly recognised and differentiated from a southern group of mussels. Within the northern group there is a high degree of genetic homogeneity, presumably resulting from high levels of gene flow among northern populations. Within the southern group there is some degree of differentiation between populations on the east and west coasts of the South Island, but the southern group as a whole is more homogeneous within itself than it is by comparison with the northern group. The location of the genetic discontinuity at  $\sim 42^{\circ}$ S is consistent with major hydrological features in the region, such as upwelling (which may move larvae offshore and away from suitable habitat) and strong coastal currents. Based on mitochondrial DNA variation data there is evidence of such coastal features acting as a barrier to gene flow, because one mitotype found in the southern group at a frequency of  $\sim 20\%$  is not found at all in the northern group. The three different studies, using different molecular approaches and applied to both mitochondrial and nuclear genomes, all indicate that despite the continuous distribution of Greenshell<sup>TM</sup> mussels throughout New Zealand, there is a profound genetic difference between the northern and southern stocks. Such genetic-based stock differences are likely to be associated with fitness differences.
- (2) The genetic difference that exists between northern and southern stocks can be used as a signal to track the movement of stocks from Kaitaia (in the northern group) to Stewart Island (in the southern group). Note that at present it is not possible to track the movement of Kaitaia spat to other northern stock sites such as the Coromandel Peninsula or the Marlborough Sounds. The studies of Apte *et al.* (2003), Star *et al.* (2003) and Wei *et al.* (2009) all clearly show that the Stewart Island aquaculture population in Big Glory Bay (derived from Kaitaia spat) shows greatest similarity to the northern group as one would expect, given its northern origins. More significantly however, the wild (natural) mussel population at Horse Shoe Bay which is <20 km from the Big Glory Bay site shows intermediate affiliation with the northern and southern groups, when it should only exhibit affinity to the southern group. This intermediate status is clear evidence of the introgression of northern genes into this wild southern population. For the first time, we have clear evidence of interbreeding and successful recruitment and subsequent development of mussels of mixed northern/southern ancestry. The geographical extent and the fitness consequences of this event are unknown.</p>

## Fitness consequences arising from interbreeding of discrete stocks of Perna canaliculus

In the present context, we define a hybrid as an individual of mixed genetic origin – the offspring of a cross between a northern and a southern Greenshell<sup>TM</sup> mussel.

The classical view (*e.g.* Mayr 1963; Dobzhansky 1970) has been that hybrids are less fit (*i.e.* they exhibit hybrid unfitness) than one or both parental types, and reviews of the literature (almost exclusively of the terrestrial hybridisation literature) tended to support this view (*e.g.* Barton & Hewitt 1985, 1989; Harrison 1990; Arnold 1992). Indeed, in some cases, hybrids have been viewed as evolutionary dead-ends because they were thought to be sterile, making them interesting oddities, but



nothing more than that. As outlined above, the explanation for this hybrid unfitness is that the act of hybridisation (the equal contribution of genes from two different stocks or species) has resulted in the break-up of co-adapted gene complexes, such that the hybrids are not genetically suited to the environment in which they arise or to which they recruit. However, more recent appraisals and reviews have indicated that in fact hybrids are often not less fit than the parental genotypes (e.g. Arnold & Hodges 1995; Gardner 1997). This suggests that not only are hybrids formed by occasional (or sometimes frequent) interbreeding of parental types, but that such hybrids (which would correctly be regarded as F1 (first filial generation) hybrids) are in fact capable of breeding, either with such hybrids (to produce F2 hybrids) or with one or both parental types to produce backcrosses. In turn, this has lead to the suggestion that hybrids may be an important evolutionary step towards the production of new genotypic variants, some of which may have greater fitness than the parental types for reasons such as heterosis (heterozygote advantage – because, by definition, F1 hybrids are the ultimate form of heterozygotes) or simply because they happen to be better fitted to the local environment as a serendipitous consequence of novel and previously untried genotypic combinations that out-perform parental genotypic combinations. Such F1 hybrids may be a bottleneck in the sense that they are rare, and therefore limit the production of new genotypic combinations (Arnold & Hodges 1995), but they may be nonetheless very important in promoting new evolutionary genotypes which enhance fitness of their holders within the environment under consideration.

In terms of the likely fitness consequences of interbreeding of north and south Greenshell<sup>TM</sup> mussels, there is very little published information on which to base predictions about relative hybrid fitness, and there is no clear indication from the literature about what to expect. The assessments of genetic variation by Gardner and colleagues (Apte *et al.* 2003; Star *et al.* 2003; Wei *et al.* 2009) point to the successful interbreeding of Kaitaia (northern stock) mussels with Horse Shoe Bay (southern wild stock) mussels when the former are transferred to the Stewart Island aquaculture site of Big Glory Bay. The genetically intermediate identity of the Horse Shoe Bay population between the northern and southern stocks strongly suggests that not only have northern genes introgressed into this southern population, but that hybrids and individuals of mixed ancestry (*i.e.* possible backcrosses) are reproductively active and that holders of novel genotypic combinations are surviving. The exact fitness status of such mussels of mixed ancestry is presently unknown, but is certainly worthy of investigation. Beyond this, it is presently not possible to say anything definitive about the genetic and fitness consequences of the mixing of the two stocks.

It is worth reiterating that our genetic knowledge of stock differences and their geographic locations is only as good as the information provided by the genetic markers. To date, we know of two distinct stocks and we have evidence of interbreeding and introgression of northern mussels at one wild southern site in Stewart Island. The present generation of genetic markers does not allow us to identify genetic differences between Kaitaia mussels and those at aquaculture sites in what is presently recognised as the northern group -i.e. the Coromandel Peninsula and the Marlborough Sounds. It must be born in mind that genetic differences may exist between the mussels of these different regions, but at present it is not possible to detect this. Thus, just because there are presently no apparent differences between these geographically distinct areas in terms of mussel genetic variation does not mean that no such differences exist. It simply means that we cannot detect them. Advances in molecular techniques and a new generation of molecular genetic tools (*e.g.* whole genome



sequencing and the identification of coding versus non-coding DNA sequence differences, and/or single nucleotide polymorphisms (SNPs) across the genome) will clarify this situation.

## What can we learn from other examples?

This section provides two different perspectives of the possible consequences of the movement, mixing and interbreeding of genetically distinct stocks of Greenshell<sup>TM</sup> mussels in New Zealand.

## Scenario I

Perhaps the best known, and most worrying example of aquaculture enhancement and its profound negative effects on wild populations in terms of decreased fitness involves the salmon industry from the western United States of America, in particular the Snake and Columbia Rivers (reviewed by Knudsen & MacDonald 1999). Salmonids such as sockeye salmon (Oncorhynchus nerka) have a highly developed homing instinct (philopatry) and return to their natal lakes, streams or rivers to breed after several years at sea. There is therefore often low or no interbreeding between fish from different natal regions, and over many generations this has resulted in high levels of genetic adaptation to the localised environment (i.e. the development of specialised co-adapted gene complexes which are highly environment-dependent) which promotes reproductive isolation, even among fish populations within a single lake (Quinn 1985; Quinn & Dittman 1990; Ramstad et al. 2004). The last 100 years has seen most wild salmonid populations in the western United States of America (and elsewhere) come under huge pressures from increasing fishing activity, increasing habitat degradation (e.g. increased stream sedimentation), dams being built which prevent the movement upstream of fish to their breeding grounds, and the loss of many streams and rivers as water is taken for crop irrigation (e.g. Costello et al. 2003). Cumulatively, these and other events have led to the reduction or extinction of localised breeding populations. To counter this, hatchery-based breeding programmes were set up to enhance wild populations. Millions of fry are bred each year and released into streams and rivers with the hope of enhancing or at least buffering wild fish populations against further losses. However, it has become apparent that the mass production of hatchery-produced fry has not had the desired effect. Typically such fish are produced from a relatively narrow genetic base (the parental stock is too small) and the fry are not genetically adapted to any particular localised environment. In short, hatchery production of fry tends to result in the breeding of fish which have high fitness in the hatchery (*i.e.* exhibit co-adapted gene complexes suited to the hatchery), but low fitness in the real world. Most worrying however, is the interbreeding of hatchery-produced fish with wild fish, and the associated impact of introgression of genes from hatchery-produced populations into wild populations which acts to decrease the mean fitness of the wild population. Thus, it may be argued that the hatchery-breeding programme has actually exacerbated the very problem that it was set-up to solve. Wild fish populations now have even lower fitness than before because the introgression of genes from hatchery-produced fish has reduced the mean fitness of wild populations.

This salmonid example is clearly not identical to the present Greenshell<sup>TM</sup> mussel situation. The primary difference is that the salmon fry are hatchery-produced (usually from wild stock), whereas the Greenshell<sup>TM</sup> mussel spat which are moved around New Zealand are wild caught (*i.e.* not hatchery produced). Nonetheless, the salmonid example provides an important lesson in the value of genetic variation which is an adaptation to the local environment, and why it is important that it not be disrupted. The concern for New Zealand's Greenshell<sup>TM</sup> mussels is that such disruption might be able



to happen here. The interbreeding of northern and southern stocks of mussels as a consequence of the movement of Kaitaia spat around the country during the last ~25 years may lead to an outcome here in which one or more wild (native) populations of mussels has decreased fitness as a consequence of the introgression of northern genes into southern stocks. While evolutionary theory suggests that the genes of less fit individuals will be removed over time from a population because, by definition, they exhibit decreased fitness compared to the genes of other individuals, it is however possible for such genes to be maintained in the population in at least two different ways. First and most likely, the constant influx of northern genes as a consequence of the mass movement of Kaitaia spat may be sufficient to overwhelm the putative removal of such genes from the local (southern) population by the process of natural selection. Second, the introgressed northern genes may be maintained in the recipient southern population because fitness differences between the different genotypic combinations are low or are not subject to natural selection challenges in the present environment. Subsequent changes may expose the true extent of the loss of mean fitness within the population. An example of this sort of case is disease resistance, and the diluting of the genes which confer such

## Scenario II

The literature pertaining to hybridisation in the sea has been reviewed by Gardner (1997). A number of different conclusions were reached in this review, the most important of which (in the present context) is that the fitness consequences of hybridisation cannot be generalised. On the basis of the data in the more than 100 papers reviewed by Gardner (1997), some generalities emerge (see below), but there is no single statement that can accurately predict the likely fitness consequences of hybridisation within any one genus. Each instance has to be treated on a case by case basis, as summarised below. In all cases, the study-specific references are given in Gardner (1997).

- Examples of marine animal hybrid unfitness are plentiful and include slower rates of development (echinoderms, molluscs), decreased fertilisation success (crustaceans, echinoderms), reduced fecundity or complete sterility (crustaceans, echinoderms, flatfish), increased mortality (echinoderms, molluscs), increased susceptibility to gonadal neoplasia (molluscs), increased morphological variability (echinoderms), lower body size or weight (echinoderms), highly skewed sex ratios (fish), and higher rates of parasitism (fish, molluscs).
- 2. Hybrids are often reported to be morphologically intermediate between the parental types, reflecting the equal contribution of the two differentiated parental genomes. Other examples of intermediate hybrid fitness include development stability and developmental rate (echinoderms, molluscs), growth rate (molluscs), body size (cetaceans, echinoderms), fertility and fecundity (crustaceans, echinoderms, molluscs), viability (crustaceans, molluscs), chromosomal structure (molluscs), allozyme thermostability (molluscs) and resistance to parasites (molluscs).
- 3. In a smaller number of cases it has been reported that hybrids have increased fitness when compared to both parental types. Examples include feeding ability (echinoderms), growth rate (fish, molluscs), fecundity (fish), and longevity (molluscs).
- 4. In summary, hybridisation in the sea results in decreased individual fitness (either narrow sense with effect only on reproductive success, or in a wider sense beyond immediate reproductive success) about as frequently as it results in intermediate or increased hybrid fitness compared with parental types. Thus, hybrids are not uniformly unfit.

One final point is worthy of mention in the context of individual fitness, especially as it might be reported from field-based assessments. This point is almost always over-looked and may only receive attention in the context of laboratory or hatchery-based assessments of hybrid fitness. In this scenario, the fieldworker may successfully estimate relative fitness of individual mussels on the shore, but without realising that only a very small subset of hybrid individuals is actually being assessed. This is because it is theoretically possible for a very large proportion of hybrid mussel larvae to be formed from the union of gametes from the two parental types, but for the vast majority of these larvae to be unfit in the sense that they carry genotypic combinations which are non-viable. Such individuals are therefore selected against before they can recruit to the shore population. Thus, they are never seen by the fieldworker who has no knowledge of the massive selection experienced by the majority of hybrid individuals. What the fieldworker observes on the shore is the very small subset of hybrid mussels which carry successful combinations of genes (within the nuclear genome and/or as a combination of the nuclear and the mitochondrial genomes) which permit survival and may even confer some form of hybrid superiority. Thus, while field-based assessments of broad-sense fitness are very important, it is equally important to carry out hatchery-based determinations of fertilisation success and larval development and mortality rates to a time beyond metamorphosis and up to the size/age of successful settlement in the field. In this way the researcher can be confident of having most accurately quantified hybrid fitness across the full range of life history stages. If this sort of scenario does indeed occur when northern and southern Greenshell<sup>TM</sup> mussel stocks interbreed, then it will result in massive wastage of gametes from the local wild stock. This may cause problems later on in terms of diminished recruitment by local mussels which may open up space on the shore for other species and result in a long-term shift in the local ecological balance.

# Recommendations

Given the fact that the mass transfer of *P. canaliculus* spat from Kaitaia to several locations in New Zealand has been going on now for at least two decades, and that this is viewed as being an activity of considerable economic benefit to New Zealand's Greenshell<sup>TM</sup> mussel industry, it seems highly unlikely that this transfer will stop (this would be the course of action under the precautionary principle). Based on this assumption, the following recommendations are advanced to help improve knowledge about the fitness consequences (if any) of the interbreeding of *P. canaliculus* stocks resulting from the mass transport of mussel spat from one region (most usually Kaitaia, but it could be elsewhere) to another. These recommendations are not solely focused on the most obvious case of northern and southern stock interbreeding (in Stewart Island), but are framed in the larger geographic context to include the transfer of, for example, northern Kaitaia spat to other northern regions including, but not limited to the Coromandel Peninsula and the Marlborough Sounds.

- Samples of wild mussels should be collected at various spatial scales (<1 km, 1-10 km, 10-100 km) from regions with a history of receiving mussel transfers from other regions. The most obvious examples will be the Coromandel Peninsula, the Marlborough Sounds and Stewart Island, although increasingly, other regions too may be affected. Samples should be tested with the best set of genetic markers available at the time to ascertain the extent (if any) of introgression of northern genes into southern wild populations.</li>
- 2. Field-based assessments of fitness (growth rate, longevity, reproductive output, parasite counts, estimates of disease *etc etc*) should be conducted at recipient sites, as well as at wild sites at various distances from the donor site (see Gardner 1994 and Gardner & Thompson 2001) to

quantify individual fitness components. Such field-based work must be complemented by molecular assessments of mussel status (multi-locus genotype).

- 3. Samples of wild mussels should be collected as soon as possible from donor and recipient locations and archived. Such collections should be carried out routinely on a 5-year basis. Collections should be of ~100 mussels per site, and samples should be stored in alcohol (100% ethanol). Such archive material will be particularly valuable for subsequent analysis with new generations of markers, as these are developed. Once analysed, the archived material will provide historical estimates of the rate of spread (if any) of introgressed genes within each region.
- 4. Hatchery-based assessments of hybridisation rates between parental mussels from different donor regions should be conducted to quantify interbreeding success at all stages in the life-history of the animals, at least up to settlement stage. This is the only sure way to quantify individual fitness from fertilisation up to settlement, and will provide an indirect estimate of "gamete wastage" of native wild mussels (*i.e.* gametes lost to unsuccessful hybridisation).

Until these combinations of approaches are carried out to quantify individual mussel fitness at various sites there is no way to know just what impact (if any) that the mass transfer of spat around the country is having on the mean fitness of local stocks. Many of these projects would be highly suited for PhD candidates to conduct and will not be that expensive to run. There is no short-cut to finding the answer to what is, from several different perspectives, a fascinating question. The answer to this question is important to industry because of the possible long-term erosion of localised genetic variation which may provide critical new genotypic combinations for a hatchery-based breeding programme, as well as to regional authorities and the general public because of the implications for local marine communities.