# Review of the Ecological Effects of Intertidal Oyster Aquaculture

Cawthron Report No. 1275 June 2007

# Review of the Ecological Effects of Intertidal Oyster Aquaculture

Barrie Forrest
Ibrahim Elmetri
Kim Clark

Prepared for

Northland Regional Council

Cawthron Institute
98 Halifax Street East, Private Bag 2,
Nelson, New Zealand.
Ph. +64 3 548 2319,
Fax. + 64 3 546 9464
www.cawthron.org.nz

Reviewed by: Robyn Dunmore        M Taylor

Robyn Dunmore        Mike Taylor

Recommended citation:
Forrest BM, Elmetri I, Clark K. 2007. Review of the Ecological Effects of Intertidal Oyster Aquaculture. Prepared for Northland Regional Council. Cawthron Report No. 1275, 25p.

# EXECUTIVE SUMMARY

## OVERVIEW OF ISSUES

The Cawthron Institute (Cawthron) was commissioned by Northland Regional Council (NRC) under the Foundation for Research, Science and Technology Envirolink scheme to carry out a review of literature on the ecological effects of intertidal oyster aquaculture in New Zealand. The purpose is to provide information that will allow NRC to determine whether, and to what extent, the effects of the 130 oyster farms under its jurisdiction should be monitored, to enable better management of aquaculture and the environment in the region.

Our review indicates that, other than a field investigation of seabed impacts in Mahurangi Harbour, little is known about the actual effects of oyster farming in New Zealand. However, relevant knowledge is provided by desktop studies of the effects of proposed oyster farms, experience with other forms of aquaculture in New Zealand, and overseas studies of oyster farm effects. Collectively, these sources of information reveal key areas of actual or potential risk from intertidal oyster cultivation as:

### Effects on the seabed

- Seabed habitat change resulting from the biodeposition of organic-rich and fine-grained oyster faeces and pseudofaeces.
- Accumulation of shell litter, debris and fouling organisms, and aggregation of predators or scavengers.
- Altered seabed topography because of altered water flows.
- Physical disturbance from farm operations.
- Shading from farm structures and crop.
- Contaminant inputs from treated timber used for farm structures.

### Effects on the water column

- Altered hydrodynamic conditions.
- Depletion of food sources, especially phytoplankton, for other organisms.
- Effects on zooplankton and eggs or larvae of marine animals.
- Alteration to nutrient cycling, reduction in dissolved oxygen levels and alteration to water clarity.

### Other effects

- Habitat creation by farm structures.
- Biosecurity issues relating to the spread of diseases, parasites and other pests.
- Effects on fish, seabirds and marine mammals.

Using a risk-based approach, the ecological significance of each of these issues was evaluated in relation to three criteria:

(i) the magnitude of impacts, which includes both the likelihood and consequences of actual or potential effects;

(ii) their spatial extent from site-specific to regional scales; and

(iii) their duration in terms of the length of time impacts would continue if farming operations were ceased and farm structures removed.

Our analysis according to these criteria revealed the following:

1. Biosecurity issues relating to the spread of pest organisms emerged as having the greatest ecological significance, consistent with views on oyster farming risks from overseas studies. By comparison with all other risk categories, the spread of pest organisms by oyster farming activities can occur at regional scales, potentially leading to ecologically significant and irreversible changes to coastal ecosystems unless effective management strategies are put in place.

2. Seabed impacts were determined to be the second most significant ecological issue. Seabed effects can be reasonably pronounced beneath oyster farms, but are highly situation-specific, meaning that their wider ecosystem significance depends on the scale of oyster farming 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). Seabed effects appear to extend no more than a few tens of metres from the perimeter of the farmed area and are likely to be reversible (should farming be discontinued) over time scales of months to years.

3. For the range of remaining issues, we considered ecological significance to be relatively minor, although there is limited knowledge about many of these. It is possible, therefore, that unrecognised cumulative effects could have already occurred from oyster farm development in New Zealand, or could arise in the future, for example: (i) in situations of high intensity oyster farming (e.g., if there are enclosed embayments dominated by oyster racks); or (ii) because of high site-specific ecological values. Without a knowledge of baseline pre-farm conditions and subsequent changes, most of the water column effects and wider ecosystem impacts would be difficult, if not impossible, to determine retrospectively.

## IMPLICATIONS FOR MONITORING IN NORTHLAND

Although the general effects of oyster farming are known, and their ecological significance can be evaluated, it is evident from our assessment that there are many knowledge gaps and areas of uncertainty. Furthermore, the nature of oyster farming impacts in the Northland context are not known. Such impacts will depend on the intensity of farming, flushing characteristics of the environment, and the sensitivity and values of adjacent habitats. It was beyond the scope and budget of this report to understand where the greatest ecological risks might occur from oyster farming in Northland in relation to these site-specific factors.

In this respect, it is premature to make comprehensive recommendations for monitoring of oyster farm effects. The development of monitoring programmes should, among other things, be based on a clear rationale for why monitoring is needed, where monitoring is undertaken (what sites), what is being measured (what indicators), at what intensity (e.g., qualitative or quantitative), and at what frequency (e.g., seasonal, annual, etc). Furthermore, monitoring results should ideally be interpreted in relation to environmental ‘bottom lines’ that reflect ‘acceptable’ levels of impact (e.g., based on recognised guidelines for environmental quality, or agreed to amongst stakeholders).

We recommend, therefore, that NRC first considers further investigation to gather site-specific knowledge about oyster farm effects in the Northland region, so that the need for monitoring (or not) can be established. Based on our risk evaluation, we suggest that the focus of this should be on understanding biosecurity risks and effects on the seabed, with the general scope of such investigations outlined in our report. Acquisition of knowledge for many of the other issues where uncertainty is high (e.g., water column effects, effects on higher trophic level animals) will require understanding of complex ecosystem processes, many of which occur beyond the immediate environment of the cultivation area (e.g., changes to phytoplankton, zooplankton, and nutrient regimes). Progress with understanding these types of issues will probably be slow, and require fundamental coastal ecosystem research.

Finally, we note that decisions regarding monitoring and ecological assessment that are made in relation to oyster farming in Northland would ideally be made in relation to other sources of environmental risk to estuarine systems in the region, so that the risks posed by oyster farming were placed in context. Risk-based methods are available for such a purpose, which can be applied in a defined area (e.g., an estuary) or across multiple regions. Such approaches provide a defensible basis for developing plans for research and monitoring, and for prioritising monitoring effort according to the greatest sources of risk.

# TABLE OF CONTENTS

# LIST OF FIGURES AND TABLES

# 1. INTRODUCTION

Pacific oysters (*Crassostrea gigas*) are grown on the intertidal flats of sheltered estuaries world-wide (Kaiser et al. 1998). They are endemic to Japan, and were first observed in New Zealand in Northland in 1971 (Dinamani 1971), probably following inadvertent introduction via shipping mechanisms such as ballast water discharge or hull fouling. Intertidal cultivation of Pacific oysters began, in favour of the native rock oyster, in the mid-1970s. There are now more than 200 farms in New Zealand covering almost 1000 hectares (Forrest and Blakemore 2002), almost all being located in estuaries of the Auckland and Northland regions (Figure 1). Key cultivation areas include the Whangaroa, Mahurangi, and Kaipara Harbours, as well as the Coromandel and the 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, to which the oysters are attached using sticks, baskets or cages (Figure 2).

**Figure 1** Pacific oyster cultivation area in northern New Zealand (Photo: R. Creese).

**Figure 2** Pacific oyster cultivation racks in Mahurangi Harbour (Photo: B. Forrest).

The Cawthron Institute (Cawthron) was commissioned by Northland Regional Council (NRC) under the Foundation for Research, Science and Technology Envirolink scheme to carry out a review of literature on the ecological effects of intertidal oyster aquaculture in New Zealand. The purpose is to provide information that will allow NRC to determine whether and to what extent the effects of the 130 oyster farms under its jurisdiction should be monitored, to enable better management of aquaculture and the environment in the region. To date there has been only one field study of intertidal oyster culture impacts in New Zealand (Forrest 1991). The Forrest (1991) work focused on ecological effects to seabed habitats in Mahurangi Harbour, and pertinent information from this work has recently been published (Forrest and Creese 2006). A broader range of potential ecological risks was recently discussed in relation to proposed oyster and mussel farm developments in Kaipara Harbour (e.g., Gibbs et al. 2005; Hewitt et al. 2006), although these studies primarily involved desktop assessments rather than acquisition of new information on impacts. Nonetheless, based on the small amount of literature for New Zealand, it is apparent that:

(i) The nature and magnitude of effects from oyster farms are similar to that described for oyster aquaculture overseas.

(ii) The broad interactions of oyster farms with the environment are similar to other forms of aquaculture, and especially mussel farming.

Hence, in this review we discuss the known effects of oyster farming in New Zealand within the wider context of other relevant knowledge from New Zealand and overseas. In particular we: (i) discuss the key ecological issues associated with intertidal oyster aquaculture; (ii) identify issues of most significance using a qualitative risk-based approach; and (iii) discuss management and monitoring requirements.

# 2. ECOLOGICAL EFFECTS OF OYSTER FARMS

## 2.1. Overview

Oyster farming, like any other shellfish aquaculture, occupies coastal space, and hence has the potential to conflict with many other uses and values (MAF 1974; Elliot 1989; DeFur and Rader 1995; Simenstad and Fresh 1995). It can have a range of adverse effects on the coastal environment either directly or indirectly, such as loss of natural character, obstruction to navigation, modification of recreational and aesthetic values, and changes to the natural environment and its associated values (Kaiser et al. 1998; Read and Fernandes 2003). In this report we restrict our discussion to effects on the natural environment from oyster farms and on-site operations. However, we recognise that ecological effects may also arise in the short-term from farm construction, and in relation to other aspects of farming operations such as off-site spat catching and product processing.

The literature on oyster farm impacts is dominated by accounts of effects on the seabed beneath culture areas (e.g., Ito and Imai 1955; Kususki 1981; Mariojouls and Sornin 1986; Castel et al. 1989; Nugues et al. 1996; Spencer et al. 1997; De Grave et al. 1998; Kaiser et al. 1998; Forrest and Creese 2006). Seabed impacts tend to be restricted to the immediate vicinity of the site, and typically extend no further than a few tens of metres or less from the perimeter of the culture area (Forrest and Creese 2006). They arise primarily from: the deposition of organic-rich particulates produced by the cultured oysters, the drop off of oysters and build-up of shell litter, changes to water flows and patterns of sediment accretion or erosion, and physical disturbance from farm operations. Oyster farms also have a number of other potential ecological effects, such as creation of a reef habitat for fouling organisms, and the associated spread of pest species. With new developments there has also been increasing interest in the carrying capacity of estuaries and harbours for oyster culture. In this respect scientific discussion has revolved around the effects of suspended particulate food (especially phytoplankton) depletion by the oyster crop and its implications for other organisms, as well as the less tractable wider ecosystem implications and effects on higher trophic level animals arising from functional changes caused by oyster farms. These issues, among others, are discussed below.

## 2.2. Effects on the seabed

### *2.2.1. Biodeposition*

Oysters derive their nutrition by filtering and processing suspended particulate matter (SPM), including detritus, inorganic particles (e.g., fine sediment), and plankton from the water column. Hence an oyster farm can be considered as a stationary biological filter that concentrates SPM from sea water as it flows through the culture, and produces waste particles in the form of faeces and pseudofaeces. The latter are mucus-bound aggregates of particles that have been filtered from the water column by oysters but which are not ingested. These wastes (generally referred to as 'biodeposits') are heavier than their constituent particles, hence readily settle on the seabed beneath culture areas (Haven and Morales-Alamo 1966; Kusuki

1981; Mitchell 2006). The deposited particles are organic-rich and consist of a substantial proportion of fine sediments (i.e., silt and clay), hence seabed sediments beneath oyster culture areas can be organically enriched and more fine-textured than sediment in surrounding areas (Forrest and Creese 2006). Organic enrichment leads to enhanced microbial activity that can result in oxygen depletion in the sediment, qualitatively evident beneath oyster farms as a mild ‘rotten egg’ smell of hydrogen sulphide and a black colour throughout most of the sediment profile. Quantitatively these coarse indicators can be measured as a decrease in redox potential within the sediment (Forrest and Creese 2006).

Organic enrichment, together with alterations to sediment grain size characteristics, can lead to associated ecological consequences. Typically these are evident as a displacement of large-bodied organisms (e.g., heart urchins, brittle stars, large bivalves) and the proliferation of small-bodied disturbance-tolerant ‘opportunistic’ species (e.g., nematodes and other marine worms). Under conditions of strong enrichment there can also be an associated decline in species diversity as is often evident in subtidal suspended culture of bivalves (e.g., Mattsson and Lindén 1983; Kaspar et al. 1985; Grant et al. 1995) and finfish (see review by Forrest et al. 2007a). In fact, the latter study highlights that the effects of enrichment beneath finfish cultures can be so pronounced that the seabed becomes uninhabitable. By comparison, the study by Forrest and Creese (2006) in Mahurangi Harbour reveals relatively mild effects that are typical of that described for oyster cultivation overseas (e.g., Kususki 1981; Mariojouls and Sornin 1986; Nugues et al. 1996; Spencer et al. 1997; De Grave et al. 1998; Kaiser et al. 1998).

The magnitude of effects from biodeposition will depend primarily on oyster stocking density and biomass, in relation to the flushing characteristics of the environment. The level of biodeposition for a given stocking density, and the assimilative capacity of the environment, may also vary seasonally in relation to factors such as water temperature (Forrest 1991). The relative role of these different attributes has not been quantified for oyster farms. In the case of intertidal culture, the capacity of the environment to assimilate and disperse farm wastes will be mainly attributable to water current speeds and wave action. Increased flushing from currents and waves will reduce biodeposition and increase oxygen delivery to the sediments, thus allowing for more efficient mineralisation of farm wastes (e.g., Findlay and Watling 1997).

Negligible enrichment effects from oyster farms in Tasmania have been attributed to low stocking densities and adequate flushing (Crawford 2003; Crawford et al. 2003). Similarly, experience with salmon farming in New Zealand and overseas shows that well-flushed sites have depositional ‘footprints’ that are less intense but more widely dispersed than shallow, poorly flushed sites (Forrest et al. 2007a). Recovery of seabed communities from the depositional effects of oyster farms is likely to be relatively rapid if farming ceases. Based on literature for mussel and salmon farms, conceivable time scales of recovery range from a few months in well-flushed areas where effects are minor (e.g., Brooks et al. 2003), to a few years in poorly flushed areas where moderate/strong enrichment effects occur (Mattsson and Lindén 1983; Hopkins et al. 2004).

### 2.2.2. *Accumulation of shell litter, debris and associated organisms*

The accumulation of shell litter, debris (e.g., pieces of oyster stick), and fouling or epibenthic organisms beneath growing racks are among the most visible effects of oyster farms (Forrest 1991; Figure 3). The range of material beneath racks reflects drop-off of cultured oysters, wild oysters and fouling organisms (see Section 2.4.1) that have established on racks and are de-fouled or slough off. Epibenthic organisms (e.g., seastars) can aggregate in response to these effects (e.g., to the external food source). Presumably hard surfaces like shell may also provide a habitat for fouling organisms.

Some of these effects are likely to be intermittent, for example shell drop-off may be exacerbated during harvesting; or depend on the type of cultivation system, for example, oyster sticks are likely to produce more crop drop off and lead to more debris than baskets. Similarly, the extent of fouling accumulation (and related effects like predator aggregation) will depend on the extent to which structures become fouled, and the extent of natural drop-off or active defouling. Other effects such as the accumulation of oyster shell, oyster sticks and other inorganic debris (e.g., calcareous shells or tubes of fouling organisms) may persist for many years after the cessation of farming (Figure 3), representing a relatively long-term change in habitat structure.

**Figure 3** Shell litter and sticks from abandoned racks (Photo: B. Forrest).

### 2.2.3. *Topographic changes*

Changes in seabed topography have been described beneath oyster farms in Mahurangi Harbour (Forrest and Creese 2006) and elsewhere (Ottmann and Sornin 1982; Everett et al. 1995). This can reflect either erosion/scouring or build-up of sediment beneath and between oyster racks. Although sedimentation rates are elevated directly beneath culture racks (Mariojouls and Sornin 1986; Sornin et al. 1987; Nugues et al. 1996), the effects on seabed topography appear more related to changes in hydrodynamic conditions caused by the rack structures (Forrest and Creese 2006). Sediment build-up to the top of racks can occur at New Zealand sites where rack alignment is perpendicular to tidal currents (Handley and Bergquist 1997; Figure 4). In such instances oyster leases have become un-useable and farming abandoned, with shell litter and debris still evident many years later as shown in Figure 3.

**Figure 4** Sediment accumulation beneath racks (Photo: B. Forrest).

### 2.2.4. Physical disturbance

In relation to the Mahurangi Harbour study, Forrest and Creese (2006) concluded that physical disturbance, for example from barges and from farm workers walking along racks (Figure 5), probably had a strong influence on the biological effects observed. The importance of physical disturbance has also been noted for intertidal oyster cultivation areas elsewhere (De Grave et al. 1998). Although there may be a mild enrichment effect beneath cultivation areas, physical disturbance is conceivably equally important as a source of impact, and perhaps more important where enrichment is negligible. However, the relative importance of these two effects would be difficult to quantify without a rigorous experimental approach. Time scales of recovery from physical disturbance are likely to be a matter of months to a few years.

**Figure 5** Farm operations are a source of physical disturbance beneath oyster racks (Photo: B. Forrest).

### 2.2.5. *Shading*

Direct impacts on the seabed could, under certain conditions, arise from shading by farm structures. This could reduce the amount of light reaching the seafloor, with implications for growth, productivity, survival and depth distribution of ecologically important primary producers such as benthic microalgae, macroalgae or seagrass (e.g., Hewitt et al. 2006).

In the context of overseas studies that report negligible effects on seagrass beneath oyster farms (e.g., Crawford 2003), we can infer that shading effects in such cases are of little significance. A number of other studies, however, have described adverse effects on seagrass beneath oyster racks and suggested shading as a possible cause (e.g., Everett et a. 1995). To our knowledge, however, the relative importance of shading vs other sources of seabed impact has never been conclusively established, and to do so would require targeted manipulative experiments. Shading effects are nonetheless theoretically possible (Hewitt et al. 2006), and 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 biodeposition) are minimal.

### 2.2.6. *Contaminant inputs*

Wooden oyster racks are constructed from treated timber, hence have the potential to leach trace contaminants such as copper, chromium and arsenic. These contaminants are likely to bind to sediments after their release and be deposited locally, with overseas studies describing elevated concentrations in seabed sediments immediately adjacent to treated piles (e.g., Weis et al. 1993). However, the release of contaminants from treated timber in seawater is reported to decrease over time (e.g., Brooks 1996; Breslin and Adler-Ivanbrook 1998), and sediment binding is likely to reduce the potential for accumulation in oysters or toxic effects on sediment-dwelling biota. Nonetheless, in the apparent absence of any information that describes this issue in relation to oyster farming, contaminant accumulation and associated toxicity cannot be discounted. It would be relatively straightforward and inexpensive to collect and analyse sediment samples from beneath oyster racks, and compare contaminant concentrations to ANZECC (2000) guidelines to ascertain whether this is an issue that warrants more thorough investigation.

## 2.3. Water column effects

### 2.3.1. *Overview*

As aquaculture expands globally and as the seabed effects of different types of bivalve and finfish cultivation become better understood, water column issues are receiving increasing attention. Early studies of suspended oyster culture overseas revealed adverse water column impacts that were related to excessive organic enrichment of the seabed. The level of seabed enrichment was such that oyster culture areas become ‘self-polluting’ as a result of oxygen depletion in the overlying water and the associated production of hydrogen sulphide at toxic concentrations (Ito and Imai 1955). The effects of intertidal oyster farming on water quality in New Zealand estuaries appear to be unknown, but we suggest that significant degradation is highly unlikely to occur given the minor to moderate levels of seabed enrichment that have been documented (Forrest 1991; Forrest and Creese 2006). Adverse water quality effects from oyster farming and other forms of aquaculture are more likely where farms are over-stocked and located in poorly flushed environments (Kusuki 1981; Wu et al. 1994; La Rosa et al. 2002). This can be avoided by appropriate site selection, and by ensuring that farm structures are configured in a way that has a minimal effect on flushing processes.

In relation to the latter point, it is recognised that there is likely to be some degree of attenuation of water currents, and alteration of patterns of water movement, within oyster farm areas (Gouleau et al. 1982; Nugues et al. 1996; Gibbs et al. 2005; Hewitt et al. 2006). This is likely to have some effects on flushing characteristics and associated fluxes of materials through farm areas. Such physical changes may also lead to effects on sediment erosion and accretion (see above), and influence other ecological processes such as patterns of planktonic larval dispersal and colonisation (Hewitt et al. 2006). However, in relation to a proposed oyster farm development in Kaipara Harbour, Hewitt et al. (2006) regarded the ecological significance of these types of effects as relatively minor. The water column issue typically given most consideration in relation to bivalve aquaculture is the role of filter-feeding by shellfish crops in depletion of suspended particulate matter (SPM) and, to a lesser extent, alteration of the distribution and cycling of nutrients. In relation to both issues the filter-feeding role played by fouling organisms associated with shellfish cultures (Section 2.4.1) may be functionally important (e.g., Mazouni et al. 2001; Mazouni 2004), although the literature tends to focus on the role of the shellfish crop in isolation, as we briefly outline below.

### 2.3.2. Food depletion

As noted in Section 2.2.1, SPM includes detritus, inorganic particles (e.g., fine sediment), and plankton (with the latter comprising phytoplankton and zooplankton). As well as organisms that spend their entire life-cycle in the plankton, zooplankton may also include temporary planktonic life-stages of invertebrates and fish, such as eggs and larvae (Gibbs et al. 2005; Hewitt et al. 2006). While depletion of zooplankton (e.g., direct consumption of fish eggs), or effects on zooplankton as a result of reduced food availability (i.e., phytoplankton depletion), are recognised as ways in which shellfish farms may affect coastal and estuarine food webs (Gibbs et al. 2005), knowledge of actual effects appears non-existent, and the literature on these issues focuses on phytoplankton depletion alone.

In New Zealand and overseas, discussion of phytoplankton depletion in relation to mussel farming and bivalve polyculture has been a much debated (e.g., Carver and Mallet 1990; Inglis et al. 2000; Hayden et al. 2000; Gibbs et al. 2002; Nunes et al. 2003; Gibbs et al. 2005), and has often been based around consideration of production carrying capacity within bays (i.e., the farm or stocking densities at which harvests are maximised). Although comparable research for oyster farms has not been conducted, the principles from mussel farming are relevant, and Gibbs et al. (2005) discuss a number of indicators that can be used to describe the general role of bivalves in controlling phytoplankton dynamics. We can deduce from such work that the role of oyster farms in removing phytoplankton from the water-column will be situation-specific and seasonally variable (Gibbs et al. 2005; Zeldis 2005). If there are sufficient densities of oysters cultivated in a region, then they can control and limit the standing stock of phytoplankton in the water-column (Gibbs et al. 2005). In fact, because of the filtration capacity of oysters (and some other bivalve species), there is much interest worldwide in the artificial enhancement of oyster populations as a means of controlling excessive phytoplankton densities in eutrophic estuaries (Newell 2004).

Theoretically, extreme levels of SPM filtration may affect not only oyster crops, but also natural populations of filter-feeders and the wider ecosystem. Such affects conceivably occur as a result of food depletion, and through alteration in SPM size spectra and phytoplankton species composition, and hence the type and quality of food available to consumers (Pietros and Rice 2003; Hewitt et al. 2006). However, such effects have never been conclusively documented in New Zealand, and periods of reduced production in shellfish aquaculture have often been attributed to larger scale processes (e.g., climatic fluctuations). Predictions made for a proposed oyster farm in South Kaipara Harbour (104 ha) suggested that some level of phytoplankton depletion was likely down-current of culture areas (e.g., Hewitt et al. 2006). However, calculations by Gibbs et al. (2005) suggested that a total aquaculture area of > 400 ha in South Kaipara Harbour (for both mussels and oysters) would be unlikely to exert significant control over SPM (especially phytoplankton) dynamics.

### 2.3.3. Nutrient cycling and related effects

The effects of oyster farming on nutrient cycling are complex and linked to the issue of food depletion discussed above. Influences on nutrient regimes from oyster farms are determined by processes involving filter-feeding and nutrient excretion, sediment remineralisation of nutrients from particulate organic matter, and loss of nutrients through oyster harvest; all of which are influenced by an array of environmental characteristics such as water temperature, water clarity, and flushing processes (Newell 2004; Porter et al. 2004). Depending on local hydrodynamics, an oyster farm will trap and concentrate or remove a portion of nutrients from the estuary ecosystem. For example, in a semi-enclosed estuary, oyster farms may effectively retain nutrients that would otherwise be exported to the sea. Some of these nutrients will be incorporated into oyster biomass, some will settle to the seabed beneath the farm as biodeposits, and some released again in a dissolved form as plant nutrients such as ammonium (Kaspar et al. 1985; Christensen et al. 2003). Release of dissolved nutrients can occur directly via excretion by the oyster stock, or indirectly via re-mineralisation and release from organically-enriched sediments.

The production of dissolved nutrients and subsequent effects on algal production involve complex processes that are 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 stimulating phytoplankton production (e.g., Pietros and Rice 2003). Conversely, because filter-feeding by oysters removes SPM from the water column, it can lead to locally increased water clarity in some circumstances. In turn this can allow increased production of seabed microalgae and seagrass, thereby reducing the flux of dissolved nutrients to the water column, and hence reducing phytoplankton production (Newell 2004; Porter et al. 2004). In relation to oyster farm development in New Zealand, alterations to nutrient cycling have been recognised, but not regarded as a significant ecological issue (Hewitt et al. 2006). Nonetheless, it must be highlighted that nutrient cycling is influenced by complex environmental relationships, hence any predictions about effects should be treated with caution.

## 2.4. Wider ecological effects

### *2.4.1. Habitat creation*

Marine farms and other artificial structures provide a three-dimensional reef habitat for colonisation by fouling organisms and associated biota. Such structures can support a considerably greater biomass and density of organisms than adjacent natural habitats (e.g., Dealteris et al. 2004), and it is now well recognised that the assemblages that develop on artificial structures can be quite different from those in adjacent rocky areas (Glasby 1999; Connell 2000), and can comprise a diverse assemblage of macroalgae and filter-feeders such as ‘sea squirts’ (Hughes et al. 2005). Hence, several studies have highlighted the possible ‘beneficial’ role played by artificial structures within the ecosystem in terms of increasing local biodiversity, enhancing coastal productivity, and compensating for habitat loss from human activities (e.g., Ambrose 1994; Hughes et al. 2005). These types of ecological roles are recognised for seabed oyster reef habitats (Perterson et al. 2003), but are not well understood for elevated oyster cultures systems. Presumably, however, the ecological role of culture systems will be comparable.

### 2.4.2. Biosecurity risks and pest organisms

Internationally, the role of aquaculture (especially the oyster industry) in the spread of diseases and pest organisms has long been recognised (Perez et al. 1981; Bourdouresque et al. 1985; Wasson et al. 2001; Leppäkoski et al. 2002; Hewitt et al. 2004). In New Zealand, biosecurity risks from diseases and pests have been evaluated to a limited extent in relation to Pacific oysters (Diggles et al. 2002) or oyster farming activities (Forrest and Blakemore 2002; Taylor et al. 2005).

In relation to disease issues, the aquaculture of oysters theoretically has the potential to lead to the introduction or development of problems with diseases and parasites, however this is unlikely to represent an ecological risk in New Zealand at present. A review by Diggles et al. (2002) reports several parasites or pathogens associated with Pacific oysters, most of which are globally ubiquitous and primarily appear to be a risk to oyster production. These include herpesvirus, which infects oyster larvae and spat, and various species of flatworm and mud-worm (e.g., Handley and Bergquist 1997). Wider ecological effects arising from a prevalence of these diseases and parasites in high densities of cultured oysters have not been reported, and expert assessment suggests that such risks are negligible (Dr S. Webb, Cawthron, pers. comm.).

More significant in the New Zealand context is the potential for oyster farm activities to spread other pest organisms, especially biofouling species that can be invasive both on artificial structures and in natural habitats. Marine farms can provide reservoirs for the establishment and subsequent spread of pest organisms, reflecting the fact that suspended structures (and associated shellfish crop) provide ideal habitats that allow some species to proliferate at high densities (e.g., Carver et al. 2003; Lane and Willemsen 2004; Coutts and Forrest 2007). The association of pest organisms with oyster farm structures in New Zealand has never been explicitly evaluated, although there are a number of examples where pest organisms have been recorded at high densities, such as the occurrence of the sea squirts *Styela clava* and *Eudistoma elongatum* on oyster farms near Auckland and in Northland, respectively (Coutts and Forrest 2005; P. Stratford, Biosecurity New Zealand, unpubl. report). Similar examples occur in South Island aquaculture regions such as prolific infestations of the sea squirts *Ciona intestinalis* and *Didemnum vexillum*, and the Asian kelp *Undaria pinnatifida* in some areas (Stuart 1997; Forrest and Blakemore 2002; Coutts and Forrest 2005, 2007).

The spread of pest species from infested structures at local scales (e.g., within bays) is primarily driven by natural dispersal mechanisms, however, spread across large areas or between regions typically occurs via inadvertent transport with human activities. For example, infested structures deployed at a marine farm (e.g., sticks, shellfish spat), or temporarily associated with it (e.g., vessels), may be transferred to other localities as part of routine aquaculture operations. There is a high likelihood that associated fouling organisms will survive where such transfers occur without the application of treatments to reduce biosecurity risks (Forrest et al. 2007b). For such reasons, the oyster industry developed and implemented management procedures for inter-regional oyster transfers when the biotoxin-producing micro-alga *Gymnodinium catenatum* was reported from northern spat catching areas. Similarly, for new proposals such as the Kaipara developments, the industry has supported the development of biosecurity management plans that aim to minimise biosecurity risks from oyster farming activities. To our knowledge, however, such approaches are not implemented across the industry as a whole.

Biosecurity risks from oyster farming pests will be most significant when: (i) target pest organisms are dispersed by oyster farming activities into regions or habitats that are optimal for their establishment and where they do not already exist, and (ii) oyster farming activities are the primary mechanism for the spread of target pests. Clearly, if a pest organism is already present in the new habitat, or is likely to spread there regardless of oyster aquaculture, for example by natural dispersal or via non-aquaculture vectors (e.g., recreational vessels), then the incremental risk posed by oyster farm operations may be negligible. On the other hand, even low intensity oyster farming has the potential to spread fouling pests, leading to ecological consequences at non-local (e.g., regional) scales. This is in contrast to most other issues discussed in this report for which effects are relatively localised, meaning their broader ecological significance is more clearly related to the intensity and geographic scale at which oyster farming is undertaken.

A final point to note is that Pacific oysters themselves are a non-indigenous species, and are regarded by some people as a pest. Populations cultivated on farms will conceivably contribute to the establishment and spread of wild populations, with high density oyster infestations often evident in natural habitats of estuaries where oysters farming occurs (B. Forrest, pers. obs.). Many of the concerns expressed regarding wild Pacific oysters relate to effects on amenity values; for example, oysters have sharp edges and reduce the appeal for walking and other recreational activities in localities where they establish (Hayward 1997). Based on the many overseas studies highlighting the functional role of Pacific oysters and the ecological effects of oyster reefs (e.g., Bernard 1974; Gottlieb and Schweighhofer 1996; Hosack et al. 2006), we would expect that wild Pacific oysters will result in significant ecological changes in habitats where they establish. These may include ‘beneficial’ effects such as enhanced diversity and abundance of biota that are associated with the three-dimensional habitat provided by dense oyster aggregations (e.g., Thomsen and McGlathery 2006 and references therein).

### 2.4.3. *Effects on fish, seabirds and marine mammals*

Effects on fish from marine aquaculture are not well understood in New Zealand, but a number of potential issues are recognised. For example, direct effects could include alteration of essential fish habitat through the deposition of shell litter and biodeposition of particulate matter, or fish could be adversely affected through trophic interactions (e.g., alteration of plankton composition and food availability). Similarly, there has been debate in New Zealand over the role of cultured shellfish in consumption of fish eggs (Gibbs et al. 2005). From a more positive perspective, on the other hand, marine farms and other artificial structures are recognised as providing shelter, habitat complexity and a food source for fish, and the aggregation of various fish species around such structures is well recognised (Relini et al. 2000; Morrisey et al. 2006). In this regard, fish associations have been described in New

Zealand studies relating to mussel farms (Gibbs 2004; Morrisey et al. 2006), but do not appear to have been considered for oysters. Hence, while the above types of effects and interactions may be possible in the case of oysters, virtually nothing is known of their significance.

Effects on seabirds have been considered for mussel and finfish farming developments in New Zealand, and some of the discussion that has arisen from this debate has relevance to oyster culture. New resting space afforded by racks may attract, and possibly benefit, some seabird species. Some predatory seabirds, such as common shag species, gannets and gulls, may benefit from food sources provided by any small pelagic fish species (e.g., juvenile yellow eyed mullet, mackerel) that are attracted to farm structures. A study that examined possible effects on King shags from the development of a large mussel farm concluded that concerns regarding entanglement, and avoidance of feeding grounds from increased boat traffic were largely unfounded (Lalas 2001). The issue of displaced or degraded feeding habitat has also been recognised for King shags in relation to mussel farm development, with concerns that mussel farms would adversely affect seabed food sources such as flounder (Butler 2003). For oyster farms, it has been recognised that adverse effects could arise due to the displacement of habitat and food sources (Kaiser et al. 1998). On the other hand, Griffen (1997) suggests that the habitat enhancement provided by seabed oyster reefs (see Section 2.4.1 above) may benefit some bird species (e.g., herons and other foraging birds) by providing an additional food supply. Conceivably suspended culture methods could have comparable effects, but we are unaware of any studies investigating such possibilities.

The potential effects of intertidal oyster farming on marine mammals (seals, dolphins and whales) are unknown. While indirect effects (e.g., because of effects of oyster farming on marine mammal food sources) are theoretically possible their likelihood is probably very low at the present level of oyster farming in New Zealand. In terms of direct effects, previous studies describing adverse interactions between aquaculture and marine mammals have highlighted entanglement and habitat exclusion as key issues (e.g., Kemper and Gibbs 2001; Kemper et al. 2003; Lloyd 2003). For oyster farms, whether such effects are possible does not appear to have been considered but, among other things, would require knowledge regarding marine mammal habitat use in relation to the distribution and location of oyster farms.

# 3. MONITORING OF ECOLOGICAL EFFECTS

## 3.1. Synthesis of issues and evaluation of ecological risks

The purpose of our review is to provide NRC with guidance on whether and to what extent the effects of the oyster farms under its jurisdiction should be monitored. Our review indicates that, other than a field investigation of seabed impacts in Mahurangi Harbour, little is known about the actual effects of oyster farming in New Zealand. Nonetheless, the Mahurangi study, together with desktop assessments, experience with other forms of aquaculture in New Zealand, and overseas knowledge, means that the key ecological issues are well recognised and their significance can be evaluated. Such an evaluation can be used to provide guidance to NRC on important knowledge gaps, or significant ecological risks, that may justify further investigation of effects, or ongoing monitoring to document changes over time. To evaluate the relative ecological significance of the issues outlined in Section 2, we have used a qualitative risk-based approach in which we have scored ecological risks from intertidal oyster farming in relation to three criteria:

(i) the magnitude of impacts, which includes both the likelihood and consequences of actual or potential effects;

(ii) their spatial extent from site-specific to regional scales; and

(iii) their duration in terms of the length of time impacts would continue if farming operations were ceased and farm structures removed.

Numeric scales for these categories were used as indicated in Table 1, which were modified from a generic process set out in the joint Australian/New Zealand Standard 4360 on Risk Management (HB 203:2000). Criteria for ranking the level of knowledge and certainty about effects are also included in Table 1. A numeric relative ranking for the overall significance of the various issues was calculated as:

*magnitude (i.e., likelihood x consequences) x spatial extent x duration*

Qualitative scores were then assigned to these values as follows:

$\leq 5$ = very low; 6-10 = low; 11-15 = moderate; $> 15$ = high

The results of this evaluation, with summary qualitative scores to indicate relative ecological significance, are given in Table 2[1]. The ranking in Table 2 should be regarded as a guide only, in that it is derived from expert opinion and is sometimes based on limited information. Furthermore, actual levels of risk will depend on many site-specific factors such as the intensity of farming in a given area, the sensitivity of the receiving environment, and the extent to which mitigation is possible. Table 2 would ideally be populated by a consensus process involving a wide group of experts and stakeholders. Nonetheless, for present purposes the

---

[1] Effects on marine mammals were not considered in the risk-based assessment in Table 2, because information is non-existent and the magnitude of potential effects cannot be easily judged against the criteria in Table 1.

**Table 1** Criteria used to rank relative ecological significance and uncertainty in Table 2. Note that likelihood was based on weightings shown (i.e., 0.2 - 1.0) rather than a 1 - 5 score.

| Score | 1 | 2 | 3 | 4 | 5 |
|---|---|---|---|---|---|
| Knowledge and certainty | Based on perception only | Perception and related information from similar activities | Limited information on effects of activity | General effects of activity known | Specific effects of activity well known |
| Likelihood | Rare (0.2) | Unlikely (0.4) | Moderate/possible (0.6) | Likely/probable (0.8) | Almost certain (1.0) |
| Consequences | Negligible | Minor | Moderate | Major | Catastrophic |
| Spatial extent (from site) | Site-specific (< 500m) | Local area (500m - 5 km) | Regional (> 5 km) | NA | NA |
| Duration | Short-term (< 1yr) | Medium-term (1 - 5 yrs) | Long-term (> 5 yrs) | NA | NA |

evaluation facilitates general understanding of the ecological significance of the various issues in a relative sense, although we suggest that only major differences in risk scores are meaningful (i.e., small differences in scores should be disregarded).

Table 2 shows that biosecurity issues relating to the spread of pest organisms receive the highest mean risk score. 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 biggest problems associated with aquaculture in estuaries (deFur and Rader 2003). The reason is that, by comparison with all other risk categories, the spread of pest organisms by oyster farming 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 a significant risk in reality depends on a number of different factors. For example, such transfers may represent a low ecological risk if the associated pest organism cannot survive in the new region, if it is already present, or if it is likely to spread to the recipient region irrespective of oyster industry activities (e.g, by natural dispersal or other vectors such as fouled recreational vessels). Furthermore, it is important to recognise that mitigation strategies may be developed to minimise biosecurity risks, such as proposed by Taylor et al. (2005) in relation to Kaipara Harbour oyster farm developments.

Seabed effects from biodeposition and physical disturbance received the second highest rankings (Table 2). These are the more obvious effects of oyster farms, and are reasonably well understood, even though their importance relative to each other is unclear. In general, seabed effects can be reasonably pronounced but highly site-specific, meaning that their wider

**Table 2** Summary of scores from qualitative assessment of actual and potential risks of intertidal oyster farming. Note that the scores are relative and should be considered as only a general guide. Actual ecological significance will depend on a range of site-specific factors (see text for details).

| Ecological issue and stressor | Comment | Knowledge & certainty | Ecological significance | | | | | |
|---|---|---|---|---|---|---|---|---|
| | | | Likelihood (A) | x Consequence (B) | Magnitude (C = A x B) | Spatial extent (D) | Duration (E) | Relative rank (C x D x E) |
| *EFFECTS ON SEABED* | | | | | | | | |
| Biodeposition/enrichment (from faeces & pseudofaeces) | Moderate change to seabed sediments and assemblages likely, but effect localised and reversible in medium term | 4 | 1 | 3 | 3 | 1 | 2 | 6.0 |
| Physical disturbance | Moderate change to seabed sediments and assemblages likely, but effect localised and reversible in medium term | 4 | 1 | 3 | 3 | 1 | 2 | 6.0 |
| Shell litter and debris accumulation | Localised physical alteration of habitat, which may persist for many years (e.g., from shell litter build-up) | 3 | 0.8 | 2 | 1.6 | 1 | 3 | 4.8 |
| Shading | Possible localised effect that could lead to effects on primary producers in the medium term | 1 | 0.6 | 3 | 1.8 | 1 | 2 | 3.6 |
| Contaminant inputs | Leaching of timber treatment contaminants likely to decrease over time, with sediment binding likely to reduce toxicity | 2 | 0.8 | 2 | 1.6 | 1 | 2 | 3.2 |
| Biofouling accumulation | Habitat alteration from biofouling drop-off possible, but effect depends on nature and extent of fouling | 3 | 0.6 | 2 | 1.2 | 1 | 2 | 2.4 |
| Altered seabed topography | Changed hydrodynamic conditions may alter patterns of sediment erosion/accretion | 4 | 0.8 | 2 | 1.6 | 1 | 3 | 4.8 |
| *EFFECTS ON WATER COLUMN* | | | | | | | | |
| Phytoplankton depletion and community change | Possible depletion/alteration of food to filter-feeders, through depletion/alteration of phytoplankton composition | 3 | 1 | 2 | 2 | 1 | 1 | 2.0 |
| Alteration to nutrient cycling | Effects vary with environmental conditions, farm stocking, and sediment enrichment | 3 | 1 | 2 | 2 | 1 | 1 | 2.0 |
| Altered hydrodynamic conditions | Retarded water flows may reduce flushing and alter fluxes of materials | 3 | 1 | 2 | 2 | 1 | 1 | 2.0 |
| Reduced dissolved oxygen | Dissolved oxygen could be depleted because of over-stocking, excessive organic enrichment, or inadequate water flushing | 2 | 0.2 | 3 | 0.6 | 1 | 1 | 0.6 |
| Zooplankton depletion and community change | Possible effects on zooplankton, including effects on eggs and larvae of marine invertebrates and fish | 1 | 0.6 | 2 | 1.2 | 1 | 1 | 1.2 |
| Alteration to water clarity | Water clarity could increase due to suspended particulate matter depletion, or decrease as a result of sediment resuspension | 2 | 0.6 | 2 | 1.2 | 1 | 1 | 1.2 |
| *OTHER EFFECTS* | | | | | | | | |
| Spread of fouling pests | Regional-scale, permanent effects possible, for which risk can be mitigated but not avoided | 4 | 0.6 | 4 | 2.4 | 3 | 3 | 21.6 |
| Habitat creation and effect on fish | Farm structures provide habitats for fouling organism, and may promote the aggregation of finfish and other marine animals | 3 | 0.6 | 2 | 1.2 | 1 | 1 | 1.2 |
| Effects on seabirds | Effects on seabirds could occur through food web effects, and habitat provision/alteration | 1 | 0.2 | 2 | 0.4 | 3 | 3 | 3.6 |
| Diseases and parasites | Oyster culture may lead to increase in disease or nuisance species, but this is unlikely to have ecological effects | 3 | 0.2 | 2 | 0.4 | 3 | 3 | 3.6 |

ecosystem significance depends on the scale of oyster 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). Seabed effects appear to extend no more than a few tens of metres from the perimeter of the farmed area and are likely to be reversible (should farming be discontinued) over time scales of several months to a few years.

There are a range of remaining issues in Table 2 for which we consider ecological significance is likely to be relatively minor, based on our discussion in Section 2, although there is limited knowledge about many of these. It is possible, therefore, that unrecognised cumulative effects could have already occurred from oyster farm development in New Zealand, or could arise in the future, for example: (i) in situations of high intensity oyster farming (e.g., if there are enclosed embayments dominated by oyster racks), or (ii) because of high site-specific ecological values. Without a knowledge of baseline pre-farm conditions and subsequent changes, most of the water column effects and wider ecosystem impacts described in Table 2 would be difficult, if not impossible, to determine retrospectively.

## 3.2. Implications for monitoring

Although the general effects of oyster farming are known, and their ecological significance can be evaluated, it is evident from the assessment above that there are many knowledge gaps and areas of uncertainty. Furthermore, the nature of impacts in the Northland context are not known. Such impacts will depend on site-specific conditions relating to the intensity of farming, flushing characteristics of the environment, and the sensitivity and values of adjacent habitats. It was beyond the scope of this report to understand where the greatest ecological risks might occur from oyster farming in Northland in relation to these site-specific factors.

In this respect, it is premature to make comprehensive recommendations for monitoring of oyster farm effects. Monitoring involves ongoing assessment of change in environmental conditions over time and is primarily conducted to ensure that environmental quality is maintained at an ‘acceptable’ level. The design of monitoring programmes should, among other things, have a clear rationale for why monitoring is needed, where monitoring is undertaken (what sites), what is being measured (what indicators), at what intensity (e.g., qualitative or quantitative), and at what frequency (e.g, seasonal, annual, etc). Furthermore, monitoring results should ideally be interpreted in relation to environmental ‘bottom lines’ that reflect ‘acceptable’ levels of impact (e.g., based on recognised guidelines for environmental quality, or agreed to amongst stakeholders). These types of approaches are becoming increasingly common in environmental monitoring and management in New Zealand, for example in relation to mussel spat-catching (Hopkins and Robertson 2002), salmon farming (Hopkins et al. 2004), and large coastal discharges (Forrest et al. 2004).

We recommend, therefore, that NRC first considers further investigation to gather site-specific knowledge about oyster farm effects in the Northland region, so that the need for monitoring (or not) can be established. Based on the risk evaluation in Table 2, we suggest that the focus of this should be on understanding:

1. **Biosecurity risks:** The emphasis here should be on gaining an understanding of the actual and potential pests that threaten estuarine values, the likelihood that identified pests will establish in Northland oyster growing areas, the oyster farming pathways by which such pests could be spread, the significance of those pathways relative to other sources of risk (e.g., recreational vessels), and the feasibility of management. This may lead to monitoring in the form of surveillance programmes (e.g., passive surveillance by the oyster industry) for the early detection (and then management) of target pest species, among other management strategies.

2. **Effects on the seabed:** Investigations should seek to understand the range of effects in Northland estuaries in relation to farming factors (e.g., stocking levels, farm size and age, farming method), and environmental factors (e.g., flushing characteristics and receiving environment values). This could be conducted as a one-off study, from which a decision could then be made on the merits of ongoing monitoring. One of the benefits of such a study is that it would guide NRC in the management of aquaculture by identifying the types of areas or management practices that allow oyster farming to be carried out with minimal impact. Such knowledge is relevant to future development, and to the mitigation of any significant adverse effects that occur at present levels of oyster farming.

Acquisition of knowledge for many of the other issues where uncertainty is high (e.g., water column effects, effects on higher trophic level animals) will require understanding of complex ecosystem processes, many of which occur beyond the immediate environment of the cultivation area (e.g., changes to phytoplankton, zooplankton, and nutrient regimes). While modelling approaches have been taken in New Zealand to evaluate food web effects (e.g., Jiang and Gibbs 2005) the large amount of data required for such models will often limit their utility. Hence, progress with understanding some of these complex issues will probably be slow, as it will require fundamental coastal ecosystem research for which funding (e.g., from central government) is limited. On a less pessimistic note, it should be recognised that the seabed can be considered as an surrogate indicator for some of these less tractable ecosystem effects; to some extent it is reasonable to assume that if seabed impacts become increasingly pronounced or widespread, the potential for wider ecosystem effects (e.g., effects on the water column and higher trophic level animals) also increases. Conversely, a minimal level of seabed impact is probably a reasonable indication that adverse effects on the wider ecosystem (other than biosecurity issues) are unlikely.

Finally, we note that decisions regarding monitoring and ecological assessment that are made in relation to oyster farming in Northland, would ideally be made in relation to other sources of environmental risk to estuarine systems in the region, so that the risks posed by oyster farming were placed in context. This holistic approach was recently applied for mussel farm development in the Firth of Thames using a Relative Risk Model (Elmetri et al. 2005). In that approach, the relative risk to predefined endpoints (particular species and populations, and habitats) from a number of sources and stressors including agricultural land use, climate change, marine farming, fishing and urban development were investigated. The outcome of the Firth of Thames work was that relative risk was identified to all of the habitats in question from all of the stressors. An important feature of the Relative Risk Model approach is that parameter uncertainty can be explicitly addressed. Such methods can be applied in a defined region (e.g., an estuary) or across multiple regions, and provide a defensible basis for developing plans for research and monitoring, and for prioritising monitoring effort according to the greatest sources of risk.

# 4. REFERENCES

Ambrose RF 1994. Mitigating the effects of a coastal power plant on a kelp forest community: rationale and requirements for an artificial reef. Bulletin of Marine Science 55: 694-708.

ANZECC, 2000. Australian and New Zealand guidelines for fresh and marine water quality 2000 Volume 1. National Water Quality Management Strategy Paper No. 4. Australian and New Zealand Environment and Conservation Council and Agriculture and Resource Management Council of Australia and New Zealand, Canberra.

Bernard FR 1974. Annual biodeposition and gross energy budget of mature pacific oysters, *Crassostrea gigas*. Journal Fisheries Research Board of Canada 31: 185-190.

Boudouresque CF, Gerbal M, Knoepffler-Peguy M 1985. L'algue japonaise *Undaria pinnatifida* (Phaeophyceae, Laminariales) en Méditerranée. Phycologia 24: 364-366.

Breslin VT, Adler-Ivanbrook L 1998. Release of copper, chromium and arsenic from CCA-C treated lumber in estuaries. Estuarine, Coastal and Shelf Science 46: 111-125.

Brooks KM 1996. Evaluating the environmental risks associated with the use of chromated copper arsenate-treated wood products in aquatic environments. Estuaries 19: 296-305.

Brooks KM, Stierns AR, Mahnken CVW, Blackburn DB 2003. Chemical and biological remediation of the benthos near Atlantic salmon farms. Aquaculture 219: 355-377.

Butler DJ 2003. Possible impacts of marine farming of mussels (*Perna canaliculus*) on king shags (*Leucocarbo carunculatus*). DOC Science Internal Series 111, Department of Conservation, Wellington, New Zealand. 29p.

Carver CE, Chisholm A, Mallet AL 2003. Strategies to mitigate the impact of *Ciona intestinalis* (L.) biofouling on shellfish production. Journal of Shellfish Research 22: 621-631.

Carver CEA, Mallet AL 1990. Estimating carrying capacity of a coastal inlet for mussel culture. Aquaculture 88: 39-53.

Castel J, Labourge JP, Escaravage V, Auby I, Garcia ME 1989. Influence of seagrass and oyster parks on the abundance and biomass patterns of meio- and macrobenthos in tidal flats. Estuarine Coastal and Shelf Science 28: 71-85.

Christensen PB, Glud RN, Daalsgaard T, Gillespie P. 2003. Impacts of longline mussel farming on oxygen and nitrogen dynamics and biological communities of coastal sediments. Aquaculture 218: 567-588.

Connell SD 2000. Floating pontoons create novel habitats for subtidal epibiota. Journal of Experimental Marine Biology and Ecology 247: 183-194.

Coutts ADM, Forrest BM 2005. Evaluation of eradication tools for the clubbed tunicate *Styela clava*. Cawthron Report No. 1110. 48p.

Coutts ADM, Forrest BM 2007. Development and application of tools for incursion response: lessons learned from the management of a potential marine pest. Journal of Experimental Marine Biology and Ecology 352: 154-162.

Crawford C 2003. Environmental management of marine aquaculture in Tasmania, Australia. Aquaculture 226: 129-138.

Crawford CM, Macleod CKA, Mitchell IM 2003. Effects of shellfish farming on the benthic environment. Aquaculture 224: 117-140.

De Grave S, Moore SJ, Burnell G 1998. Changes in benthic macrofauna associated with intertidal oyster, *Crassostrea gigas* (Thunberg) culture. Journal of Shellfish Research 17: 1137-1142.

Dealteris JT, Kilpatrick BD, Rheault RB 2004. A comparative evaluation of the habitat value of shellfish aquaculture gear, submerged aquatic vegetation and a non-vegetated seabed. Journal of Shellfish Research 23: 867-874.

deFur PL, Rader RN 1995. Aquaculture in estuaries: feast or famine? Estuaries 18: 2-9.

Diggles BK, Hine PM, Handley S, Boustead NC 2002. A handbook of diseases of importance to aquaculture in New Zealand. NIWA Science and Technology Series No. 49, 200pp. ISSN 173-0382.

Dinamani P 1971. Occurrence of the Japanese oyster, *Crassostrea gigas* (Thunberg), in Northland, New Zealand. New Zealand Journal of Marine and Freshwater Research 5: 352-7.

Elliot M 1980. Mahurangi: problems of management. Soil and Water, April. p. 26-28.

Elliot M 2003. Biological pollutants and biological pollution - an increasing cause for concern. Marine Pollution Bulletin 46: 275-280.

Elmetri I, Gibbs M, Landis W 2005. Application of the Relative Risk Model (RRM) to investigate the risks of a proposed aquaculture development in the Firth of Thames. Prepared for Environment Waikato, Auckland Regional Council and Ministry for the Environment. Cawthron Report 1058. 50p.

Everett RA, Ruiz GM, Carlton JT 1995. Effects of oyster mariculture on submerged aquatic vegetation: an experimental test in a Pacific northwest estuary. Marine Ecology Progress Series 125: 2205-2217.

Findlay RH, Watling L 1997. Prediction of benthic impact for salmon net-pens based on the balance of benthic oxygen supply and demand. Marine Ecology Progress Series 155: 147-157.

Forrest BM 1991. Oyster farm impacts on the benthic environment: a study in Mahurangi Harbour. Master of Science thesis, University of Auckland, New Zealand. 157p.

Forrest BM, Blakemore K 2002. Inter-regional marine farming pathways for the Asian kelp *Undaria pinnatifida*. Cawthron Report No. 726. 27p. plus appendices.

Forrest BM, Creese RG 2006. Benthic Impacts of Intertidal Oyster Culture, with Consideration of Taxonomic Sufficiency. Environmental Monitoring and Assessment 112: 159-176.

Forrest BM, Gibbs M, Hatton S 2004. Doubtful Sound adaptive management programme. Cawthron Report No. 932. 13p.

Forrest BM, Keeley N, Gillespie PA, Hopkins GA, Knight B, Govier D 2007a. Review of the Ecological Effects of Marine Finfish Aquaculture: Final Report. Cawthron Report No. 1285. 71p.

Forrest BM, Hopkins GA, Dodgshun TJ, Gardner JPA 2007b. Efficacy of acetic acid treatments in the management of marine biofouling. Aquaculture 262: 319-332. Cawthron Report No. 932. 13p.

Gibbs M, Ross A, Downes M 2002. Nutrient cycling and fluxes in Beatrix Bay, Pelorus Sound, New Zealand. New Zealand Journal of Marine and Freshwater Research 36: 675-697.

Gibbs MT 2004. Interactions between bivalve shellfish farms and fishery resources. Aquaculture 240: 267-296.

Gibbs MT, Hatton S, Gillespie PA, Forrest BM 2005. A desktop assessment of potential and cumulative effects on the plankton, benthos, and water column of the proposed Aquaculture Management Areas in the South Kaipara Harbour. Cawthron Report No. 980. 33p.

Glasby TM 1999. Differences between subtidal epibiota on pier pilings and rocky reefs at marinas in Sydney, Australia. Estuarine, Coastal and Shelf Science 48: 281-290.

Gottlieb SJ, Schweighofer ME 1996. Oysters and the Chesapeake Bay ecosystem: a case for exotic species introduction to improve environmental quality? Estuaries 19: 639-650.

Gouleau D, Ottmann F, Sornin JM 1982. Repercussion de la conchyliculture sur la sedimentation. *In*: La gestion régionale des sédiments. Séminaire de Propriano, Documents No. 3, BRGM:189-198.

Grant J, Stenton-Dozey J, Monteiro P, Pitcher G, Heasman K 1998. Shellfish culture in the Benguela system: a carbon budget of Saldanha Bay for raft culture of *Mytilus Galloprovincialis*. Journal of Shellfish Research 17: 44-49.

Griffen K 1997. Eelgrass ecology and commercial oyster cultivation in Tillamook Bay, Oregon: a literature review and synthesis. Unpublished TBNEP Oyster Report, July 1997. 19p

Handley SJ, Bergquist PR 1997. Spionid polychaete infestations of intertidal Pacific oysters *Crassostrea gigas* (Thunberg), Mahurangi Harbour, northern New Zealand. Aquaculture 153: 191-205.

Haven DS, Morales-Alamo R 1966. Aspects of biodeposition by oysters and other invertebrate filter feeders. Limnology and Oceanography 11: 487-498.

Hayden B, Ross A, James M, Hadfield M, Gibbs M 2000. Carrying capacity: the way to sustainable shellfish production. Aquaculture Update 7-9.

Hayward B 1997. Introduced marine organisms in New Zealand and their impact in the Waitemata Harbour, Auckland. Tane 36: 197-223.

Hewitt CL, Willing J, Bauckham A, Cassidy AM, Cox CMS, Jones L, Wotton DM 2004. New Zealand marine biosecurity: delivering outcomes in a fluid environment. New Zealand Journal of Marine and Freshwater Research 38: 429-438.

Hewitt J, Morrison M, Spigel R, Turner S 2006. Oyster Farming in the Kaipara Harbour Proposed Aquaculture Management Area D. Ecological Assessment. Prepared for Biomarine Ltd.NIWA Client Report: HAM2006-118.

Hopkins GA, Forrest BM, Clarke M 2004. Environmental impacts of the Otanerau Bay salmon farm, Marlborough Sounds. Cawthron Report No. 824. 57p plus appendices.

Hopkins GA, Robertson B 2002. Proposed adaptive management plan for the Tasman Bay and Golden Bay spat-catching sites. Cawthron Report No. 758. 19p.

Hosack GR, Dumbauld BR, Ruesink JL, Armstrong DA 2006. Habitat associations of estuarine species: comparisons of intertidal mudflat, seagrass (*Zostera marina*), and oyster (*Crassostrea gigas*) habitats. Estuaries and Coasts 29: 1150-1160.

Hughes DJ, Cook EJ, Sayer MDJ 2005. Biofiltration and biofouling on artificial structures in Europe: the potential for mitigating organic impacts. Oceanography and Marine Biology: an Annual Review 43:123-172.

Inglis G, Hayden B, Ross A 2000. An overview of factors affecting the carrying capacity of coastal embayments for mussel culture. NIWA Client Report: CHC00/69. 31p.

Ito S, Imai T 1955. Ecology of oyster bed I. On the decline of productivity due to repeated cultures. Tohoku Journal of Agricultural Research 5: 251-268.

Jenkins RJ, Meredyth-Young JL. 1979. Occurrence of the Pacific Oyster, *Crassostrea gigas*, off the South Island of New Zealand. NZ Journal of Marine and Freshwater 13:173-174.

Jiang WM, Gibbs MT 2005. Predicting the carrying capacity of bivalve shellfish culture using a steady, linear food web model. Aquaculture 244: 171–185.

Kaiser MJ, Laing I, Utting D, Burnell GM. 1998. Environmental impacts of bivalve mariculture. Journal of Shellfish Research 17: 59-66.

Kaspar HF, Gillespie PA, Boyer IC, MacKenzie AL. 1985. Effects of mussel aquaculture on the nitrogen cycle and benthic communities in Kenepuru Sound, Marlborough Sounds, New Zealand. Marine Biology 85: 127-136.

Kemper CM, Gibbs SE 2001. Dolphin interactions with tuna feedlots at Port Lincoln, South Australia and recommendations for minimising entanglements. Journal of Cetacean Research and Management 3: 283-292.

Kemper CM, Pemberton D, Cawthorn M, Heinrich S, Mann J, Wursig B, Shaughnessy P, Gales R 2003. Aquaculture and marine mammals: co-existence or conflict? pp 208-224. *In*: Marine Mammals: Fisheries, Tourism and Management Issues. Gales N, Hindell M Kirkwood R (Eds), CSIRO Publishing.

Kusuki Y 1981. Fundamental studies on the deterioration of oyster growing grounds. Bull. Hiroshima Fish. Exper. Stn 11: 1-93.

La Rosa T, Mirto S, Favaloro E, Savona B, Sara G, Danovaro R, Mazzola A 2002. Impact on the water column biogeochemistry of a Mediterranean mussel and fish farm. Water Research 36: 713-721.

Lalas C 2001. Evidence presented for Kuku Mara Partnership, Forsyth Bay. Environment Court Hearing. Blenheim. Statements of Evidence – Volume 1. October 2001.

Lane A, Willemsen P 2004. Collaborative effort looks into biofouling. Fish Farming International, September 2004: 34-35.

Leppäkoski E, Gollasch S, Olenin S 2002. Invasive aquatic species of Europe: distribution, impacts and management. Kluwer Academic Publishers, Dordrecht, the Netherlands. 583p.

Lloyd BD 2003. Potential effects of mussel farming on New Zealand's marine mammals and seabirds: a discussion paper. Department of Conservation, Wellington, New Zealand. 34p.

MAF 1974. Oyster farming, Houhora Harbour: Environmental impact report. Ministry of Agiculture and Fisheries, New Zealand. 52p.

Mariojouls C, Sornin JM 1986. Sur exploitation et détérioration de la qualité des terrains conchylicoles: conséquences sur les systèmes d’exploitation - exemples en France et au Japon. Norois 34: 51-61.

Mattsson J, Lindén O 1983. Benthic macrofauna succession under mussels, *Mytilus edulis* L. (Bivalvia), cultured on hanging long-lines. Sarsia 68: 97-102.

Mazouni N 2004. Influence of suspended oyster cultures on nitrogen regeneration in a coastal lagoon (Thau, France). Marine Ecology Progress Series 276: 103-113.

Mazouni N, Gaertner JC, Deslous-Paoli JM 2001. Composition of biofouling communities on suspended oyster cultures: an *in situ* study of their interactions with the water column. Marine Ecology Progress Series 214:93-102.

MFish 2006. Ministry of Fisheries, Aquaculture. Retrieved 16 January 2007, from http://www.fish.govt.nz/en-nz/Commercial/Aquaculture/default.htm

Mitchell IM 2006. In situ biodeposition rates of Pacific oysters (*Crassostrea gigas*) on a marine farm in Southern Tasmania (Australia). Aquaculture, 257 (1-4): 194-203.

Morrisey DJ, Cole RG, Davey NK, Handley SJ, Bradley A, Brown SN, Madarasz AL 2006. Abundance and diversity of fish on mussel farms in New Zealand. Aquaculture 252: 277-288.

Morrisey DJ, Gibbs MM, Pickmere SE, Cole RG 2000. Predicting impacts and recovery of marine-farm sites in Stewart Island, New Zealand, from the Findlay-Watling model. Aquaculture 185: 257-271.

Newell RIE 2004. Ecosystem influences of natural and cultivated populations of suspension-feeding bivalve molluscs: a review. Journal of Shellfish Research 23: 51-61.

Nugues MM, Kaiser MJ, Spencer BE, Edwards DB 1996. Benthic community changes associated with intertidal oyster cultivation. Aquaculture Research 27: 913-924.

Nunes JP, Ferreira JG, Gazeau F, Lencart-Silva J, Zhang XL, Zhu MY, Fang JG 2003. A model for sustainable management of shellfish polyculture in coastal bays. Aquaculture 219: 257-277.

Ottmann F, Sornin JM 1982. Relationship between marine bottom elevation and various types of marine culture. Atlantica 5: 88-89.

Perez R, Lee JY, Juge C 1981. Observations sur la biologie de l'algue japonaise *Undaria pinnatifida* (Harvey) Suringar introduite accidentellement dans l'Etang de Thau. Science et Pêche 315: 1-12.

Peterson CH, Grabowski J H, Powers S P 2003. Estimated enhancement of fish production resulting from restoring oyster reef habitat: quantitative valuation. Marine Ecology Progress Series 264: 249-264.

Pietros JM, Rice MA 2003. The impacts of aquacultured oysters, *Crassostrea virginica* (Gmelin, 1791) on water column nitrogen and sedimentation: results of as mesocosm study. Aquaculture 220: 407-422.

Porter ET, Cornwell JC, Sanford LP 2004. Effect of oysters *Crassostrea virginica* and bottom shear velocity on benthic-pelagic coupling and estuarine water quality. Marine Ecology Progress Series 271: 61-75.

Read P, Fernandes T 2003. Management of environmental impacts of marine aquaculture in Europe. Aquaculture 226: 139-163.

Relini G, Relini M, Montanari M 2000. An offshore buoy as a small artificial island and a fish-aggregating device (FAD) in the Mediterranean. Hydrobiologia 440: 65-80.

Simenstad CA, Fresh KL 1995. Influence of intertidal aquaculture on benthic communities in Pacific Northwest estuaries: scales of disturbance. Estuaries 18: 43-70.

Sornin JM, Delmas D, Deslous-Paoli JM 1987. Evolutions quantitatives et qualitatives du seston dans une claire a huîtres: relation avec la sedimentation et la biodeposition. Océanis 13: 531-541.

Spencer BE, Kaiser MJ, Edwards DB. 1997. Ecological effects of intertidal Manila clam cultivation: observations at the end of the cultivation phase. Journal of Applied Ecology 34: 444-452.

Stuart M. 1997. The seasonal ecophysiology of *Undaria pinnatifida* (Harvey) Suringar in Otago Harbour, New Zealand. PhD Thesis, University of Otago. 184p.

Taylor MD, Dodgshun TJ, de Zwart E. 2005. Biosecurity management plan for a proposed south Kaipara oyster farm. Cawthron Report No. 1123, Cawthron Institute, Nelson, New Zealand. 37p.

Thomsen MS, McGlathery K 2006. Effects of accumulations of sediments and drift algal on recruitment of sessile organisms associated with oyster reefs. Journal of Experimental Marine Biology and Ecology 328: 22-34.

Wasson K, Zabin CJ, Bedinger L, Diaz MC, Pearse JS 2001. Biological invasions of estuaries without international shipping: the importance of intraregional transport. Biological Conservation 102: 143-153.

Weis JS, Weis P, Proctor T 1993. Copper, chromium, and arsenic in estuarine sediments adjacent to wood treated with chromated-copper-arsenate (CC). Estuarine, Coastal and Shelf Science 36: 71-79.

Wu RSS, Lam KS, MacKay DW, Lau TC, Yam V 1994. Impact of marine fish farming on water quality and bottom sediment: a case study in the sub-tropical environment. Marine Environmental Research 38: 115-145.

Zeldis J 2005. Magnitudes of natural and mussel farm-derived fluxes of carbon and nitrogen in the Firth of Thames. Environment Waikato Technical Report 2005/30. 42p.

# ACKNOWLEDGEMENTS

We are grateful to Nigel Keeley and Robyn Dunmore for review comments on a draft of this report.