Ministry for Primary Industries
Manatū Ahu Matua

LITERATURE REVIEW OF ECOLOGICAL EFFECTS OF AQUACULTURE

Benthic Effects

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Benthic Effects 3

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3.1 Introduction

The propensity for aquaculture to cause significant effects on the seabed (and water column) can be roughly determined by the diet and feeding mechanism of the candidate species, their waste production and culture method. The cultivation of organisms that require external feed inputs (e.g. finfish, crayfish, paua) are likely to produce more waste products than cultivation of species that do not rely on external feeds. The combination of excreted waste and uneaten feed has a relatively high potential to adversely affect the local seabed (within approximately 1 km), as is evident in the case of salmon farming (Section 3.2 and Forrest et al. 2007). 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 (Section 3.3 and Keeley et al. 2009). Nonetheless, in high density culture situations filter-feeding pressure can result in pronounced seabed effects in certain environments. The cultivation of organisms such as seaweeds (macroalgae) and sea cucumbers that function at a lower trophic level and/or utilise only dissolved nutrients and sunlight presumably leads to minimal ecological effects (Section 3.4 and Keeley et al. 2009).

3.2 Feed-added species (salmon, kingfish, hapuku)

3.2.1 Overview of seabed effects

Note: The following summary draws heavily on a review that was conducted by Forrest et al. (2007) and references contained therein. At times, information has been condensed and source references have been omitted for the sake of brevity and readability, however, they can be found in that document. Additions and amendments have been made based on new understanding or information.

Fish farms are almost invariably sited above soft-sediment habitats (as opposed to rocky habitats) and therefore the information on seabed effects relates primarily to physico-chemical and ecological changes in such areas. Most of the literature describes the effects of salmon farming, but studies for other finfish species (e.g. yellowtail kingfish, European sea bass, red sea bream) reveal that seabed impacts are similar (e.g. Karakassis et al. 1999; Rajendran et al. 1999; Mazzola et al. 2000; Yokoyama 2003). The dominant effect on the seabed arises from the deposition of faeces and uneaten feed, which leads to over-enrichment of the seabed due to the high organic content of the deposited particles. Hence, there is considered to be a high degree of transferability between the effects that have been described for salmon and those that are likely to occur for lesser known fish species, as long as the feed type and farming methods (e.g. feeding mechanisms, stocking densities) are comparable. To the best of our knowledge, the effects of finfish culture on the seabed outlined here will be applicable to both kingfish and hapuku.

The seabed effects that result from finfish farming have been described according to the scale of the resulting effects (i.e. localised and within the primary footprint or far-field wider ecosystem) as listed below. The most dominant and well-described effects concern localised seabed enrichment from biodeposits and this appropriately comprises the bulk of the discussion. Other related effects include those of biofouling drop-off and shading by structures. It should also be noted that finfish farms produce significant quantities of dissolved nutrients and, therefore, the potential exists for waterborne enrichment of the benthos.

This section has been structured according to the following main types of ecological effects:

- Organic enrichment and smothering (3.2.2.1):
  - Localised biodeposition leading to enrichment of the seabed and associated microbial processes, and chemical and biological changes (including infauna and epifauna).
  - Smothering of benthic organisms and/or changes in physical composition of sediments.
  - Widespread biodeposition leading to mild enrichment in naturally depositional areas. Potential for effects on reefs, inshore habitats and sensitive taxa.
  - Sediment contamination (copper and zinc) covered in the additives chapter (Chapter 10).
- Biofouling drop-off and debris (3.2.2.2):
  - Leading to organic enrichment and changes to physical composition of sediments.
  - Leading to aggregations of predators and scavengers.
- Seabed shading by structures (3.2.2.3).

3.2.2 Descriptions of main effects and their significance

3.2.2.1 Organic enrichment and smothering

Table 3.1: Organic enrichment due to biodeposition from feed-added aquaculture operations – localised effects.

Description of effect(s)	Feed and faecal deposition from finfish farms can change well-aerated and species-rich soft sediments in the vicinity of farm cages into anoxic (oxygen-depleted) zones that can be azoic (devoid of life) in extreme cases. Microbial decay of the waste material can dramatically alter the chemistry and ecology of the seafloor. Benthic communities can become highly enriched, infaunal diversity will be significantly reduced and extreme abundances of common opportunistic taxa may occur. Organic accumulation is less at highly dispersive sites, but the sediment chemistry and general composition will be significantly altered. Beneath finfish farms, enrichment effects are usually inseparable from those of farm-derived contaminants (e.g. copper and zinc), which is likely to be a compounding factor.
Spatial scale	*Local to bay-wide scale* – Effects most evident directly beneath the cages and exhibit a strong gradient of decreasing impact with increasing distance. The intensity and spatial extent of enrichment is highly site specific, with high flow, deep sites producing larger but more diffuse footprints. Mild enrichment can be detected out to about 100 to 1000m away from the farm, dependent on the site's dispersive properties.
Duration	*Short to long term* – Significant recovery is short term, occurring within the first few months (approximately three to 12 months) of cessation of deposition. The benthos is mostly recovered in the medium to long term, within the timeframe of months to years (estimated 5–10 years for low flow sites in New Zealand). However, if trace metals accumulate in the sediments then they may continue to retard recolonisation after the organic material is gone, in which case full recovery may take longer.
Management options	Can be partially controlled through: • careful site selection; • altering feed capacities, optimising feed management (and farm production and/or intensity) and matching farm placement and design to site; • monitoring and ongoing adaptive management. Impacts reversible upon removal of farm.
Knowledge gaps	Enrichment effects on reef biota. Comparative recovery rates at high flow sites.

* Italicised text in this table is defined in chapter 1 – Introduction.

Table 3.2: Smothering of benthic organisms by biodeposits from feed-added aquaculture operations.

Description of effect(s)	Smothering effects are closely related to enrichment effects as both are caused by elevated levels of biodeposition and, in many cases, occur concurrently and are, therefore, difficult to separate. The distinction is made because the resuspension processes that dominate highly dispersive sites tend to preclude smothering effects from accumulative deposition; however, the effects of enrichment are usually still evident. Conversely, at low flow sites, "inundation" and smothering by biodeposits are likely to contribute significantly to the effects.
Spatial scale	*Local* scale (tens to hundreds of metres from farm) – Smothering effects tend to be more localised than enrichment effects because they tend to occur at low flow, depositional sites, where biodeposits will not spread as far, compared to sites where enrichment effects are more prevalent.
Management options	Site selection.
Knowledge gaps	None identified.

* Italicised text in this table is defined in chapter 1 – Introduction.

Localised biodeposition

The microbial decay of organic waste material (predominantly feed and faeces) can dramatically alter the chemistry and ecology of the seafloor (Forrest et al. 2007 and references therein). More than 20 years of research and investigation, both within New Zealand and overseas, has consistently shown that feed and faecal deposition from finfish farms can change well-aerated and species-rich soft sediments in the vicinity of farm cages into anoxic (oxygen-depleted) zones that can be azoic (devoid of life) in extreme cases, or dominated by only a few sediment-dwelling species tolerant of the degraded conditions.

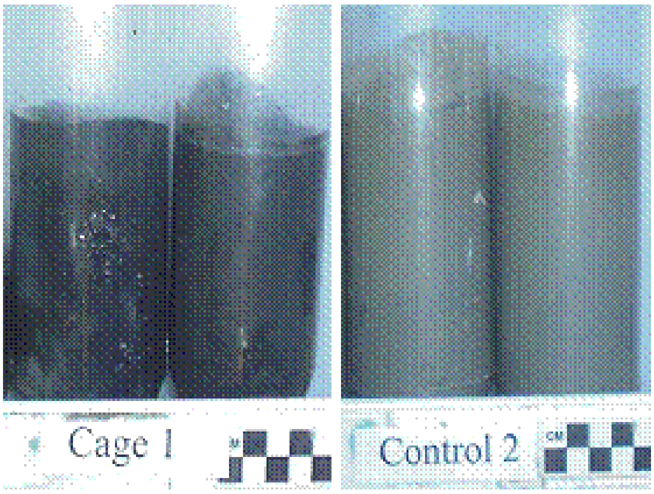
The depositional "footprint" of a typical finfish farm extends tens to hundreds of metres from the point of discharge (Brown et al. 1987; Karakassis et al. 2000; Schendel et al. 2004; Chagué-Goff & Brown 2005), often in an elliptical pattern that is skewed in the direction of prevailing currents. Effects tend to be most evident directly beneath the cage, and exhibit a strong gradient of decreasing impact with increasing distance, which is consistent with other organic enrichment gradients (see review by Pearson & Rosenberg 1978). Farm-derived particulates may disperse further than the footprint of measurable effects, as shown by a recent overseas study detecting farm wastes up to 1 km from the source (Sara et al. 2004). Such findings highlight that the seabed environment beyond the effects footprint may be exposed to farm-derived materials, but at a rate that is able to be assimilated without exhibiting any measurable ecological changes.

Excessive levels of organic enrichment directly beneath finfish farms are typically identified using a suite of different "indicators". Anoxic conditions within the sediment are evident as a strong "rotten egg" smell of hydrogen sulphide from sediment samples and a black colour throughout the sediment profile (Figure 3.1). Such conditions will typically be accompanied by visible white or cream coloured patches across the seafloor, which indicate the presence of mat-forming filamentous bacteria such as *Beggiatoa* sp. (Figure 3.2). Under extreme conditions, sediment out-gassing also occurs, which will be evident as gas bubbles emerging from the sediment surface (Iwama 1991; Hopkins et al. 2004). This gas predominantly comprises hydrogen sulphide and methane, which is formed through the process of sulphate reduction and methanogenisis in the presence of anaerobic conditions (Gowen & Bradbury 1987; Hargrave et al. 2008);

The hydrogen sulphide component of the out-gassing can adversely affect the health of fish and other fauna (Gowen & Bradbury 1987; Black et al. 1996). Under such conditions, levels of sediment organic matter and nutrients (e.g. organic carbon, nitrogen and phosphorus) are usually significantly elevated in comparison to natural sediments (Karakassis et al. 2000; Gao et al. 2005). The sediment can also be enriched with trace contaminants (e.g. zinc, copper) sourced from feed or antifouling agents. The specific effects of copper and zinc are discussed in more detail in Sneddon & Tremblay (2011) and the additives chapter (Chapter 10), but it is also relevant to note here that they are common additional stressors that occur in association with organic enrichment beneath salmon farms. As such, the ecological effects of copper and zinc are also part of, and encompassed by, assessments of benthic effects.

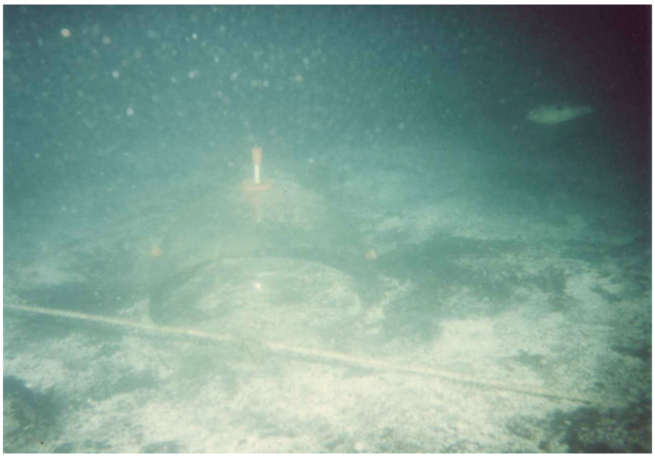
Enrichment leading to seabed sediments devoid of infauna (animals that inhabit the sediment matrix) has been described in the past for many salmon farms in New Zealand (e.g. Edwards 1988; Forrest 1996a; Chagué-Goff & Brown 2003; 2004; 2005; Hopkins et al. 2006a; 2006b; 2006c), but the development of management strategies to reduce this risk has largely been successful (e.g. Otanerau Bay Farm: Keeley et al. 2011; Ruakaka Bay Farm: Forrest et al. 2011). The rapid reduction in the severity of physico-chemical effects with increasing distance from the farm leads to an associated reduction in ecological effects. Most studies characterise ecological changes using infaunal communities (and other complementary techniques); the presence or absence, abundance and diversity of organisms that inhabit the sediments are well-recognised indicators of seabed health and enrichment status (Pearson & Rosenberg 1978; Brown et al. 1987; Keeley et al. 2012a, b).

Figure 3.1: Mud samples from beneath salmon cages in the Marlborough Sounds



Notes: Left: black anoxic sediments from beneath cages compared with brown sediments from a control site beyond the influence of the farm. Right: sediment grab sample with black sediment and faecal material (orange) evident.

Figure 3.2: Seafloor beneath salmon cages in the Marlborough Sounds showing bacterial cover (*Beggiatoa* sp.) present as a result of pronounced organic enrichment



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 enrichment effects on epibiota from finfish farms in New Zealand are not well documented, although Forrest (1996) provides one example where epibiota were observed beneath salmon cages in a well-flushed environment. Similarly, organisms such as sea cucumbers, cushion stars and snake stars have been observed aggregating under conditions of mild enrichment at New Zealand salmon farming sites (Govier & Bennett 2007a), sometimes in association with bacterial mats. These fauna tend to be displaced in situations of high enrichment, in which case they can be absent directly beneath the cages but aggregated around the perimeter where the enrichment is less intense. Epibiota may also respond to salmon farm effects other than direct deposition. For example, they may scavenge fouling biota that have fallen (or been defouled) from the farm structures.

Widespread biodeposition

Wider ecological effects from farm-derived biodeposits are possible due to resuspension processes that can transport organic particles beyond the primary footprint. However, the dilution and dispersion factors are such that distant ecological effects are usually minimal and/or difficult to detect over and above natural temporal and spatial variability. This is because much of the suspended particulate organic matter will be sufficiently diffuse that it can be naturally assimilated in the water column and/or on the seabed.

The extent to which resuspension spreads the waste material is determined by a site's, physical properties (i.e. depth and current speeds). At low flow sites very little resuspension occurs and effects are largely constrained to the local environment (Forrest et al. 2007). At high flow sites, however, the majority of the biodeposits are resuspended and exported, which promotes dilution and assimilation by the environment, and a portion may eventually be deposited in a diffuse form in neighbouring low flow areas (e.g. in blind bays). If depositional inputs are sufficiently elevated then there is potential for far-field benthic enrichment. In which case, although the magnitude of change would be very small the spatial extent could be very large.

Therefore, habitats outside of the primary footprint (e.g. ecologically important inshore communities, downstream reefs) may be affected by elevated suspended particulate loads (biodeposits) when resuspension occurs. This has the potential to negatively impact some species by creating an environment that is too turbid, blocking light (in the case of photosynthetic taxa) and potentially impeding larval settlement (e.g. Rodriguez et al. 1993; Walker 2007). Conversely, some taxa may benefit from the increased availability of organic particulates and dissolved nutrients (e.g. suspension-feeding bivalves, Teaioro 1999; Keeley 2001). This issue is particularly pertinent at high flow sites, because they tend to coincide with the physical requirements of reef communities, which often contain "potentially sensitive" or "ecologically valuable" taxa, in particular, large sessile filter feeders (e.g. hydroids, sponges) and macroalgae (e.g. kelp). However, by their very nature, high flow sites are well flushed and non-depositional and, therefore, inherently resilient to the effects of biodeposition and smothering. Direct observations of reef communities adjacent to New Zealand King Salmon farms operating in high flow areas for up to five years are yet to detect any obvious negative effects from resuspended farm-generated wastes (Dunmore & Keeley 2013).

Table 3.3: Organic enrichment due to biodeposition from feed-added aquaculture operations – widespread effects.

Description of effect(s)	Widespread but very diffuse benthic enrichment is possible outside of the primary footprint in near-by naturally depositional areas (e.g., blind bays). In most cases, the rate of deposition is likely to be low enough to be naturally assimilated. Any effects are likely to be subtle and difficult to detect. The amount of material that is exported from a farm and therefore available for deposition elsewhere is dependent on the dispersive properties of the site.
Spatial scale	*Regional* – Potentially large scale, i.e. that is, tens to hundreds of hectares. Cumulative across farms in the area.
Duration	*Short term* – Any low level enrichment is likely to be reversible within a relative short timeframe.
Management options	Site selection, system wide hydrodynamic modelling to identify potential hotspots.
Knowledge gaps	None identified.

* Italicised text in this table is defined in chapter 1 – Introduction.

3.2.2.2 Biofouling drop-off and debris

Table 3.4: Biofouling drop-off and debris from feed-added aquaculture operations leading to organic enrichment and changes to physical composition of sediments.

Description of effect(s)	Drop-off of biofouling is most obvious around the farm perimeters beneath net sides. This can occur naturally (sloughing and natural drop-off) and unnaturally (net cleaning and dropping of litter). It is thought to contribute substantially to organic enrichment in those areas. Shell material and debris can also alter the physical and chemical composition of the seabed and can affect the benthic fauna; infaunal composition can be altered and diversity enhanced by providing substrate for sessile organisms.
Spatial scale	*Local* scale – Limited to the areas directly beneath the nets and up to a few metres away. However, dispersal range will increase at deep and/or very high velocity sites, but this is still likely to be in the order of tens of metres from the cages.
Duration	*Short* to *long term* – Associated enrichment is reversible within a similar timeframe to enrichment from feed and faeces. Shell material will take longer to breakdown and revert to natural conditions. Inorganic debris (e.g. rope, cable ties) are unlikely to break down in the foreseeable future.
Management options	Natural drop-off can be partially manageable by controlling net rotations, antifouling methods, cage design and so on. Drop-off from cleaning is highly manageable by preventing in situ cleaning. Littering is manageable through industry best management practices.
Knowledge gaps	Lack of information pertaining to how much fouling drop-off contributes to benthic enrichment over and above feed and faeces deposition. Lack of information quantifying the contribution of different farm practices (e.g. in situ net cleaning) to drop-off.

* Italicised text in this table is defined in chapter 1 – Introduction.

Table 3.5: Biofouling drop-off and biodeposition from feed-added aquaculture operations leading to aggregations of predators and scavengers.

Description of effect(s)	Biofouling drop-off and elevated biodeposition can lead to aggregations of scavenging and/or predatory organisms, such as sea cucumbers, sea stars, crabs and lice. These fauna tend to be displaced under highly enriched conditions, in which case they may aggregate around the perimeter of the farm.
Spatial scale	*Local* scale – Limited to the areas directly beneath the nets and up to about 50m away.
Duration	For the duration of the farm – however this effect is reversible as mobile predators are likely to move away or starve once the food source is removed.
Management options	Amount of attractant (or food source) partially controllable through composition of farm structures and net cleaning practices.
Knowledge gaps	Very limited information regarding any possible ecological effects of predator aggregations. What happens when a farm is removed and the organisms disperse?

* Italicised text in this table is defined in chapter 1 – Introduction.

Summary

Deposition of fouling biota may also contribute to seabed enrichment. One example arises in situations where fouling organisms reach high densities on farm structures and fall to the seabed either naturally or because of deliberate defouling by farm operators. Shell material and debris can alter the physical and chemical composition of the seabed and can affect the benthic fauna (Keeley et al. 2009); infaunal composition can be altered and diversity enhanced by providing substrate for sessile organisms. The fouling biomass may intermittently be a substantial component of the organic material deposited on the seafloor, as appears to be the case when blue mussels or the invasive sea squirt *Didemnum vexillum* are removed from nets on salmon farms in the Marlborough Sounds (author's pers. obs.). In such situations, the deposited fouling biomass may exacerbate enrichment effects (at least in the short term) associated with other processes.

Table 3.6: Shading of seabed by structures on feed-added aquaculture farms.

Description of effect(s)	The presence of farm structures could reduce the amount of natural light (PAR) reaching the seabed, thereby reducing algae productivity. Changes would be most evident when situated in naturally clear water.
Spatial scale	*Local* scale – Roughly equate to two to three times the area of the structures.
Duration	For the duration of the farm – Microalgae productivity responds quickly to changes in ambient conditions, hence it would be expected that the benthic microflora would rapidly re-establish if the farm was removed.
Management options	Site selection, fine scale positioning of cages, matching feed levels to a sites physical properties and staged adaptive management.
Knowledge gaps	None identified.

* Italicised text in this table is defined in chapter 1 – Introduction.

Biofouling drop-off and elevated biodeposition can lead to aggregations of scavenging and/or predatory organisms, such as sea cucumbers, sea stars, crabs and lice. These fauna tend to be displaced under highly enriched conditions, in which case they may aggregate around the perimeter of the farm.

3.2.2.3 Seabed shading by structures

Direct effects on the seabed can arise via processes other than deposition alone. For example, shading from farm structures can reduce the amount of natural light (photosynthetically active radiation, PAR) reaching the seafloor. This in turn could reduce the productivity of ecologically important primary producers such as benthic microalgae, or beds of macroalgae or eelgrass, with a range of associated ecological effects (e.g. Huxham et al. 2006). This issue could arise if farms are located in environments of relatively high water clarity, especially in well-flushed locations where deposition effects were low. Although identified as a potential effect, no studies exist that separate the effects of shading from that of benthic enrichment; presumably because they occur concurrently and the latter is thought to be the dominant stressor. Hence, this is a site-specific issue and one that can be at least partially mitigated by site selection.

3.2.3 Factors relating to all benthic impacts

3.2.3.1 Main factors affecting the extent of seabed effects

The magnitude and spatial extent of seabed effects from finfish farms (Sections 3.2.2.1 to 3.2.2.3) are a function of a number of inter-related factors that can be broadly considered as farm attributes and physical environment attributes.

Farm attributes

Farm attributes that can affect the mass load of organic material deposited to the seabed include fish stocking density and the settling velocities of fish faeces. The latter appears to vary considerably among fish species from about 0.4–6.0 cm s^{-1}; Magill et al. 2006), and hence may influence relative deposition levels.

Other farm attributes include the types of feed and feeding systems, the feeding efficiency of the fish stock and the settling velocities of waste feed pellets. Depositional rates can also be influenced by farm waste consumption by wild fish assemblages. Clearly, it is in the interests of the fish farmer to minimise feed wastage. As well as the economic costs associated with waste feed, excessive food loss can organically enrich the seabed to a point where water column effects occur (e.g. hydrogen sulphide production) and fish health may be compromised.

The type of cage structure may also influence depositional effects through differences in fish holding capacity, which affects feed loadings and may affect feeding efficiencies. The arrangement of the cages will also obviously affect the distribution of the seabed effects. Tightly clustered steel cages will have a localised and intense footprint in comparison to a more widely distributed cluster of individual plastic circular cages. Furthermore, cage design and position may affect depositional patterns through altering the way water currents move around a farm site. Any reductions in flow will reduce waste dispersal and flushing, potentially resulting in effects that are relatively localised but also more pronounced.

Physical site attributes

The capacity of the environment to disperse and assimilate farm wastes is primarily a function of water depth and current speeds, although assimilative capacity may also vary seasonally in relation to factors such as water temperature. Water depth and current speeds affect the extent of flushing, therefore, they are the primary attributes that modify both the magnitude and spatial extent of seabed effects. Increased flushing not only reduces localised sedimentation and accumulation of organic matter, but it also increases oxygen delivery to the sediments, thus allowing for more efficient mineralisation of farm wastes (Findlay & Watling 1997). Consequently, sites located in deep water (more than 30 metres) and exposed to strong water currents (more than 15 cm s^{-1} on average) will have more widely dispersed depositional footprints with less intense enrichment than shallow, poorly flushed sites (e.g. Molina Dominguez et al. 2001; Pearson & Black 2001; Aguado-Gimenez & Garcia-Garcia 2004), Keeley et al. 2013a, b).

Contrasts in seabed effects between high and low flow environments are evident in the case of salmon farming in the Marlborough Sounds. Several existing farms in areas of weak flushing, such as Forsyth and Ruakaka Bays, have localised but quite pronounced effects (e.g. Forrest 1996; 2007b Govier & Bennett 2007a, 2007b). By contrast, at two farms in the high current environment of Tory Channel in Queen Charlotte Sound, the intensity of effects will be substantially less when subjected to comparable feed levels (Keeley et al. 2012b, 2013a).

In terms of the types of seabed effects beneath the farms, organic accumulation tends to be minimal at high flow (dispersive) sites due to the increased levels of resuspension and the exporting of particles elsewhere. This is evidenced by relatively small increases in the sediment organic content (percentage of ash free dry weight (AFDW)) beneath farms at high flow sites compared with low flow sites, where organic content can increase six-fold (Keeley et al. 2012b, 2013a). Changes to the infaunal community at high flow sites are not as obvious during the early stages of enrichment; however, they can be very pronounced and characteristically different at higher feed levels. Most notably, extreme abundances (more than 23 000 individuals/core) of opportunistic species (primarily *Capitellid* sp. and nematodes) can develop in the centre of the footprint, and natural benthic diversity tends higher and can be maintained throughout higher levels of enrichment. At low flow sites, peak abundances are usually between 2000 to 3000 individuals/core, and diversity is compromised at earlier stages of enrichment. The main ecological responses that characterise benthic enrichment at high and low flow sites are summarised in Table 3.7.

3.2.3.2 Seabed recovery

One of the ways in which the significance of human activities in coastal environments can be assessed is to consider whether they cause permanent or long-term changes, or whether adverse effects are reversible once their cause is removed. This is a pertinent question to address in the case of new farm developments and has particular relevance for the evaluation of mitigation strategies based on farm fallowing and rotation.

Fish farm studies in New Zealand and overseas indicate timescales of recovery ranging from months to years. Recovery rates are influenced by the spatial extent and magnitude of enrichment at the point of fallowing, and the flushing characteristics of the environment (Karakassis et al. 1999; Brooks et al. 2003); larger and more heavily impacted sites, or sites in areas of relatively weak currents, are expected to take longer to recover. A number of overseas studies describe partial recovery within the first three to six months after the cessation of farming (Mazzola et al. 2000; Brooks et al. 2003; Macleod et al. 2004), but complete recovery (i.e. comparable to background conditions) can take many years and is often not fully realised in the timeframe of monitoring programmes (Karakassis et al. 1999; McGhie et al. 2000; Pohle et al. 2001; Pereira et al. 2004). The process tends to involve an initial improvement in the intensity of physico-chemical effects, with a slower timescale of recovery for seabed faunal communities (Pohle et al. 2001; Brooks et al. 2004; Macleod et al. 2004). Recovery is also thought to be adversely affected by the presence of contaminants (i.e. copper and zinc) as persistence in the sediments may impede infaunal health and therefore ecological succession (see additives chapter (Chapter 10)). The large range in estimates for seabed remediation and recovery is partly due to the wide variety of criteria that has been proposed.

The best studied New Zealand example of seabed recovery is the Forsyth Bay salmon farm in the Marlborough Sounds, which was completely fallowed (all farming structures were removed) in November 2001. Prior to being fallowed, the sediments beneath the site were highly enriched, with extensive coverage of the seabed by bacterial mats, highly elevated organic levels and out-gassing at the water surface. Infaunal abundance and richness were both markedly suppressed, indicative of near-azoic conditions (Hopkins 2002; Hopkins et al 2004). Shortly after farming ceased (i.e. two to three years), there was a significant reduction in the magnitude of effects, indicated by a reduction in sediment organic content, increased species diversity and abundance, and a corresponding decrease in the number of opportunistic species such as the polychaete *C. capitata*. Conditions at the site continued to improve three to five years after fallowing, but the size of the improvements became incrementally smaller with time. Prior to reinstatement in November 2009, and eight years after fallowing, the seabed at the Forsyth Farm had recovered according to some establishment criteria, but trace effects were still evident. Most of the measured environmental variables were similar to reference sites; however, the infaunal community in previously impacted sediments (i.e. beneath the old cage site) was still different from the reference site. Hence, our best estimate of the time required for impacted low flow sites in the Marlborough Sounds to fully recover is between five and ten+ years, dependent on the situation and the selected endpoint definition. It is expected that recovery will be faster at well flushed sites due to the high levels of resuspension, oxygentation and the associated limited propensity to accumulate organic material and become excessively impacted.

3.2.3.3 Characterising and quantifying enrichment

Typical changes in environmental variables along an enrichment gradient are graphically represented in Figure 3.3, and general descriptions for seven general enrichment stages (ES) are provided in Table 3.7. This enrichment gradient has been adapted specifically for New Zealand salmon farms based on Cawthron's more than ten years' of experience monitoring farms in the Marlborough Sounds and Big Glory Bay, Stewart Island. It is based on the well-known concept of ecological succession in stressed environments (Pearson & Rosenberg 1978) that has been adopted into ecological models (e.g. Grall & Glémarec 1997) and used in benthic health indexes (e.g. AZTIs Marine Biotic Index (AMBI), Borja & Muxika 2005), and on enrichment gradients that have also been described in association with salmon farms elsewhere in the world (e.g. Macleod & Forbes 2004; Wildish & Cranston 1997). It encompasses the full range of possible effects, from pristine natural conditions (ES = 1) to extremely enriched conditions (ES = 7), characterised by sediment anoxia (without oxygen) and azoic conditions (i.e. uninhabitable by macrobiota and/or infauna). An important feature along the gradient is the stage of greatly enhanced seabed productivity, which defines ES 5 and is evidenced by extreme proliferation of one or a few enrichment-tolerant "opportunistic" species such as the marine polychaete worm *C. capitata* and nematodes. ES 5-type conditions have traditionally been recommended at the upper level of acceptable seabed effects beneath salmon farms in the Marlborough Sounds. At ES 5, the benthos is still considered biologically functional and associated with the greatest biomass and is, therefore, thought to have greatest waste assimilation capacity.

Stages beyond ES 5 are characterised by extremely impacted sediments and the collapse of the infaunal population, at which point organic accumulation of waste material may greatly increase. Species richness generally declines with increasing enrichment (as indicated by increasing organic content and sulphides, and reducing redox potential[12]); although an area of increased richness can occur in the early stages (about ES 2.5–3, Table 3.7). Such pronounced and predictable changes in the infaunal community composition mean that the effects of organic accumulation can be reliably assessed with the use of biotic indices such as the Shannon-Wiener Diversity Index and AMBI (e.g. Borja et al. 2009; Keeley et al. 2011a).

[12]Reduction potential (also known as redox potential, oxidation/reduction potential, ORP or Eh) is a measure of the tendency of a chemical species to acquire electrons and thereby be reduced. Redox potential reflects the degree of oxidation of the sediment.

Figure 3.3: Stylised depiction of changes in infaunal abundance, species richness (number of taxa), sediment organic content and sulphide and redox levels along an enrichment gradient, defined by enrichment stage (ES) 1–7

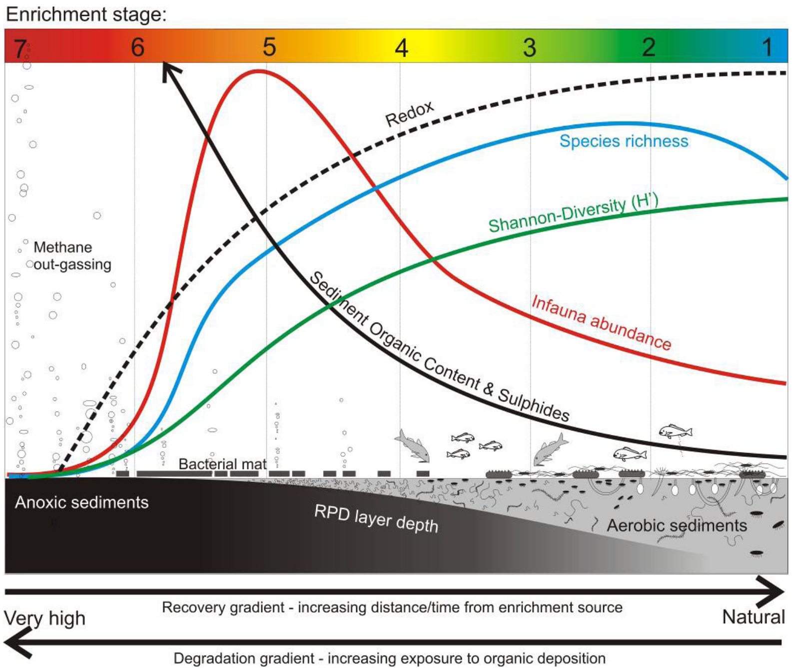


Table 3.7: General description and main environmental characteristics of Enrichment Stages (ES) 1 to 7 differentiated for low flow (LF) and high flow (HF) sites

ES	General description		Environmental indicators
1	Natural/pristine conditions.	LF	Environmental variables comparable to unpolluted unenriched pristine reference site.
		HF	As for LF, but infauna richness and abundances naturally higher (about twice that of low flow) and percentage Organic Matter slightly lower.
2	Minor enrichment. Low-level enrichment. Can occur naturally or from other diffuse anthropogenic sources. "Enhanced zone".	LF	Richness usually greater than for reference conditions. Zone of "enhancement" – minor increases in abundance possible. Mainly compositional change. Sediment chemistry unaffected or with only very minor effects.
		HF	Changes as for LF.
3	Moderate enrichment. Clearly enriched and impacted. Significant community change evident.	LF	Notable abundance increase, richness and diversity usually lower than reference site. Opportunistic species (i.e. capitellid worms) begin to dominate.
		HF	As for LF.
4	High enrichment. Transitional stage between moderate effects and peak macrofauna abundance. Major community change.	LF	Diversity further reduced, abundances usually quite high, but clearly sub-peak. Opportunistic species dominate, but other taxa may still persist. Major sediment chemistry changes (approaching hypoxia).
		HF	As above, but abundance can be very high while richness and diversity are not necessarily reduced.
5	Very high enrichment. State of peak macrofauna abundance.	LF	Very high numbers of one or two opportunistic species (i.e. capitellid worms, nematodes). Richness very low. Major sediment chemistry changes (hypoxia, moderate oxygen stress). Bacterial mat (*Beggiatoa*) usually evident. H_2S out-gassing on disturbance.
		HF	Abundances of opportunistic species can be extreme (up to ten times that of LF ES 5 densities). Diversity usually significantly reduced, but moderate richness can be maintained. Sediment organic content usually slightly elevated. Bacterial mat formation and out-gassing possible.
6	Excessive enrichment. Transitional stage between peak abundance and azoic (void of any organisms).	LF	Richness and diversity very low. Abundances of opportunistic species severely reduced from peak, but not azoic. Total abundance low but can be comparable to reference site. Percentage of organic material can be very high (3–6 times the reference site).
		HF	Opportunistic species strongly dominant, taxa richness and diversity substantially reduced. Total infauna abundance less than at sites further away from farm. Elevated organic matter and sulphide levels. Formation of bacterial mats and out-gassing.
7	Severe enrichment. Anoxic and azoic; sediments no longer capable of supporting macrofauna, with organics accumulating.	LF	None, or only trace numbers of macrofauna remain. Some samples with no taxa. Spontaneous out-gassing; *Beggiatoa* usually present but can be suppressed. Percentage of organic matter can be very high (three to six times reference site).
		HF	Not previously observed – but assumed similar to LF sites.

3.2.4 Impact mitigation and management strategies

Most of the ecological effects described above relate to seabed enrichment and stem from elevated rates of biodeposition and, accordingly, can be managed by monitoring the magnitude and spatial extent of the primary depositional footprint. The general methods for this type of monitoring are reasonably well established. However, wider ecological effects beyond the primary depositional footprint are less well understood and not as obvious, and the monitoring methods are accordingly varied and often novel. Such monitoring therefore needs to be targeted and site specific, and to also consider wider, regional scale changes, which requires spatially and temporally appropriate sampling designs.

A typical approach to establishing and then managing and monitoring marine farms in New Zealand is as follows.

Site selection: Selecting for dispersive properties (flow, depth, connectivity with larger water bodies) and broad-scale positioning to avoid potentially sensitive and/or valuable habitats (conservation areas, reefs and so on). This is usually achieved by utilising local knowledge of the region, hydrodynamic models, aerial imagery and habitat mapping.

Fine-scale positioning of cages: Mapping habitats, modelling footprints and adjusting the position of cages to optimise dispersal of wastes and minimise impacts on potentially sensitive habitats.

Matching feed levels to farm’s physical characteristics: Depositional modelling is used to predict spatial extent and magnitude of environmental effects and contrast a range of farming scenarios to inform decisions regarding optimum (sustainable) site-specific feed capacities.

Staged development/Modelling-On growing-Monitoring (MOM) approach: A conservative approach to fish farm developments involves starting at relatively low production levels, staging the development while conducting targeted ecological effects monitoring and making future expansions conditional upon acceptable environmental outcomes. This adaptive management-type approach usually requires establishing acceptable zones of effects (AZEs), in terms of distances from the farm (e.g. the Zones concept – see below). The existing farms in the Marlborough Sounds have been managed in this manner since 2003, and similar approach also underpins management in other major salmon producing countries such as Norway (e.g. The MOM system; Ervik et al. 1997; Hansen et al. 2001).

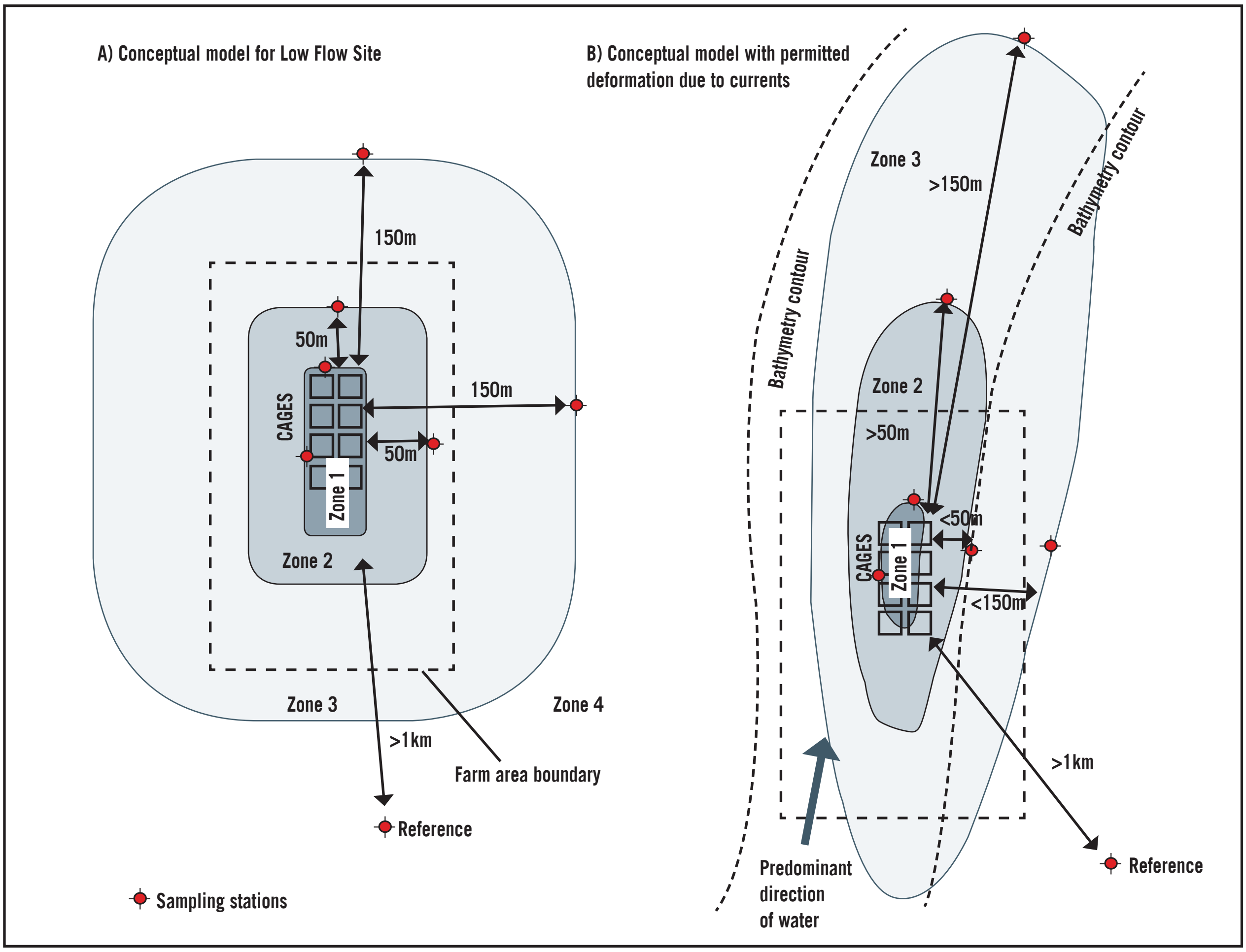
3.2.4.1 Example of spatial management: The Zones concept

A staged adaptive management approach to farm development is dependent on the establishment of clear compliance criteria against which the seabed effects associated with feed increases can be evaluated. The Zones concept provides such a framework, whereby seabed conditions are compared against pre-specified environmental quality standards (EQS) that relate to both the magnitude (or "severity") and spatial extent of effects. This basic approach has been successfully utilised for monitoring and managing existing New Zealand King Salmon farm sites since 2003.

The Zones concept is based around the spatial delineation of site-specific zones of effects as depicted in Figure 3.4. Four zones are proposed: Zone 1 encompasses a relatively small area, not much bigger than the area occupied by the cages (or in this case, encompassed by the cage area boundary), and is used to check against the maximum acceptable level of seabed effects. In the absence of strong currents (e.g. Figure 3.4-A), Zone 2 typically encompasses an area of seabed out to 50 metres away from the cages in any direction and conditions at the Zone 2/3 boundary are compared to those in Zone 1 to assess infauna peak. Similarly, Zone 3 extends out to 150 metres away, and the outer boundary (Zone 3/4 boundary) represents the accepted maximum extent of measurable ecological effects when compared against a reference site. Zone 4 represents unimpacted reference conditions and is anywhere outside of the Zones 1 to 3 (i.e. lies outside the primary footprint).

However, in the presence of strong currents, the footprints can become elongated and/or skewed around the cages and, therefore, the zones need to be permitted to deform to accommodate different site characteristics (e.g. Figure 3.4-B). At highly dispersive sites, the overall size of the footprint is typically larger than for low flow sites and the distances to the Zone 2/3 and Zone 3/4 boundaries may need to be expanded accordingly.

Figure 3.4: (A) Conceptual approach to defining seabed impact zones for typical low flow salmon farm sites, and (B) A proposed method for adapting the impact zones to the environmental conditions present at more dispersive sites



3.2.4.2 Environmental variables and quality standards (EQS)

EQS are critical to the Zones concept as they provide the quantitative criteria against which effects are assessed. EQS are measurable environmental values that are selected to reflect ecological indicators of certain stages of enrichment and are usually linked to pre-defined spatial proximities from the enrichment source (i.e. spatial zones). Globally, there is a wide array of quantitative indicator variables that have been used, including (but not limited to) sediment characteristics (e.g. particle grain size, organic content, sulphide and redox levels) and trace metal (e.g. copper & zinc) concentrations (Appendix 3.2). Perhaps the most widely used and reliable indicator of sediment condition is the state of the animals that live within the sediments (i.e. infauna). Several variables are used to describe the state of the infaunal community, from simple metrics such as total abundance, abundance of pollution tolerant taxa and the total number of taxa, to more complicated diversity indices such as the Shannon-Wiener Diversity Index, evenness measures, AZTI's Marine Biotic Index (AMBI), Benthic Quality Index (BQI) and the Infaunal Trophic Index (ITI). These latter biotic indices are particularly useful for describing enrichment gradients (Keeley et al. 2012b, Borja et al. 2009). There are also several common qualitative indicators such as the presence of *Beggiatoa* (a white mat-forming bacterium), sediment out-gassing (of methane CH_4 and hydrogen sulphate H_2S) and sediment odour – all of which can be measured on categorical scales.

Internationally, there is reasonable consistency with the types of variables that are being used, however, different countries do favour different variables (Wilson et al. 2009, refer summary tables in Appendix 3.1 and 3.2) and inter-regional validation of some biotic indices is recommended prior to general application (Keeley et al. 2012b). In terms of the actual thresholds, or EQS, international consistency is harder to find as different countries adopt different monitoring strategies with regard to spatial boundaries (or AZEs, e.g. the United Kingdom) and the EQS associated with each. Perhaps the most common types of EQS are: redox levels greater than 0 near to the farm, various sulphide concentration limits, Australian and New Zealand Environment Conservation Council (ANZECC) (2000) interim sediment quality guidelines (ISQG) levels for copper and zinc, general guidelines constraining the amount to which mono-specific dominance of opportunistic taxa can occur (i.e. abundance and diversity type measures) and prohibiting azoic conditions. The various qualitative measures listed above are also commonly used because they are relatively cheap, easy to measure and reasonably reliable.

Generic application of EQS is further complicated by the fact that, as discussed above in Section 3.2.3.1, the dispersive properties of a site influence the way the benthos responds to enrichment. In New Zealand, this is being overcome by treating the sites in two flow categories (dispersive and non-dispersive) and developing separate EQS accordingly (i.e. Table 3.7, see also Keeley et al. 2012b, 2013b). Difficulties in comparing amongst EQS are also being addressed by relating them to a standard enrichment gradient, from ES 1 (pristine) to ES 7 (azoic) (Figure 3.3). Enrichment stages greater than ES 5 are generally considered inappropriate; hence maintaining seabed conditions at or below ES 5 is a recommended management goal within Zones 1 and 2. The EQS for the boundary between Zones 2 and 3 may simply require a lower enrichment level (e.g. ES < 4) and/or that infaunal abundances are no greater than is observed for the stations beneath the cages. This criterion is based on the premise that higher abundances around the perimeter of the enrichment source indicates that post peak abundances (indicative of ES 6-type conditions) are present in the centre of the footprint. Further away, at the outer boundary of the predicted primary footprint (beyond Zone 3, Figure 3.4), the level of enrichment may be required to remain comparable relative to an appropriate reference stations (i.e. natural or background conditions), and ES less than < 3). The former ensures relative change does not occur, and the latter caters for the possibility of a sliding background; i.e. the reference site being progressively enriched. Each of the seven enrichment stages (see Figure 3.3 and Table 3.7) have been quantified using a suite of environmental indicator variables and validated for the Marlborough Sounds (author's pers. obs.). Using this process, different combinations of environmental variables can be used to quantitatively determine the enrichment stage for any sampling station.

3.3 Filter feeders (Green-lipped mussels and Pacific oysters)

3.3.1 Introduction

This section considers the ecological effects associated with farming New Zealand's two main cultivated bivalve (filter-feeding) species: green-lipped mussels (*Perna canaliculus*) and Pacific oysters (*Crassostrea gigas*). The information available reflects New Zealand's experience; therefore, studies of the effects of mussel aquaculture primarily concern long-line subtidal culture methods and studies of the effects of oyster culture are based mostly on intertidal stick or basket type farms. The following summaries draw heavily on a review of the ecological effects of non-finfish types of aquaculture (Keeley et al. 2009) and a related review on the effects of intertidal oyster culture in New Zealand (Forrest et al. 2009). At times, information has been condensed and source references may have been omitted for the sake of brevity and readability; however, they can be found in those documents. Additions and amendments have been made based on new understanding and information.

- Organic enrichment and smothering (3.3.2.1):
  - Localised biodeposition leading to enrichment of the seabed and associated microbial processes, and chemical and biological changes (including infauna and epifauna).
  - Smothering of benthic organisms and/or changes in sediment physical composition.
  - Biofouling drop-off and debris leading to organic enrichment and changes to composition of sediments.
  - Widespread biodeposition leading to a reduction in natural deposition rates.
- Changes to physico-chemical properties of the sediments (3.3.2.2).
- Changes to biological properties of the sediments (3.3.2.3).
- Effects on epibiota (3.3.2.4).
- Crop and biofouling drop-off and debris (3.3.2.5):
  - Leading to habitat creation.
- Seabed shading by structures (3.3.2.6).

3.3.2 Description of the main effects

Seabed effects from mussel and oyster farms result from the sedimentation of organic-rich, fine-grained particles (mussel faeces and pseudofaeces), and the deposition and accumulation of live mussels/oysters, mussel/oyster shell litter and other biota attached to the ropes, floats and the mussels/oysters themselves. The predominant effects on the seabed arise from the deposition of organically rich shellfish faeces and pseudofaeces (referred to as "biodeposits"), and from shell debris, which reduce flow and percolation of oxygenated water into the sediment. Although the basic process leading to enrichment is analogous to finfish farms, the rates of deposition are much lower because shellfish do not involve additional feed inputs and, accordingly, the typical level of effects are much less. Mussel and oysters 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.

3.3.2.1 Organic enrichment and smothering

Table 3.8: Organic enrichment due to biodeposition from filter-feeder aquaculture – localised effects.

Description of effect(s)	Faecal pellet and pseudofaecal production by mussels and/or oysters increases sedimentation rates under culture sites. This results in changes in sediment texture and local organic enrichment with an associated increase in oxygen consumption, increased nitrogen release rates, sulphate reduction and lowered redox potential. Increased organic loading usually results in a mildly enriched infauna. The enrichment level is generally much lower than for finfish farms, i.e. ES 2–4, Figure 3.3, Table 3.7.
Spatial scale	*Local* scale – Enrichment from mussels is usually limited to within 50m of farm structures. Oysters produce more localised enrichment (tens of metres from structures) due to depth (intertidal).
Duration	Reversible within the *medium* term to *long term*.
Management options	Site selection, reducing stocking densities.
Knowledge gaps	None identified.

* Italicised text in this table is defined in chapter 1 – Introduction.

Table 3.9: Smothering of benthic organisms by biodeposits from filter-feeder aquaculture operations.

Description of effect(s)	Excessive biodeposition in low flow sites can result in smothering of benthos, impacting the biota through a different mechanism to enrichment alone. Rates of biodeposition from filter feeders are relatively low and smothering effects are expected to be minimal in comparison to enrichment. Expected to be even less of an issue with oysters due to lower densities (per square metre) and tidal flushing.
Spatial scale	*Local* scale – Expected to be within 10 to 20 metres of farm structures for mussels and directly beneath structures for oysters.
Duration	*Medium* to *long term*.
Management options	Site selection, reducing stocking densities.
Knowledge gaps	Separating the effects of smothering from those of enrichment is unclear for mussels while the knowledge gaps for oysters have not been identified.

* Italicised text in this table is defined in chapter 1 – Introduction.

Table 3.10: Biofouling drop-off and debris from filter-feeder aquaculture leading to organic enrichment and changes to composition of sediments

Description of effect(s)	For mussels, biota and crop naturally drop to the seabed during cultivation or if lines become over crowded. Fouling organisms, including significant biomass of unwanted blue mussels, are discharged back to the seabed during harvesting. Shell material alters the physical and chemical composition of the seabed, which in turn impacts the infaunal and epifaunal communities. Diversity can increase due to increased availability of hard substrates – essentially promoting the formation of reef-type communities. Aggregations of predatory species may result. Wild oysters and associated epibiota naturally colonise structures, some of which fall to the seabed. Intertidal oyster culture also results in non-biological litter or debris in the form of sticks, baskets and cable ties. These alter the physical and chemical composition of the seabed, which in turn impacts the infaunal and epifaunal communities. Aggregations of predatory species may result.
Spatial scale	*Local* scale – Directly beneath lines (less than 10 metres away).
Duration	*Medium* to *long term* – Shell material and accumulated inorganic debris may take several years to break down.
Management options	Best management practices to minimise drop-off (i.e. avoid over-crowding, reduce discard rate of over-settlements).
Knowledge gaps	The relative contribution of biofouling (as distinct from faecal matter) to enrichment beneath farms. Break-down time of shells (mussels and oysters) in sediments.

* Italicised text in this table is defined in chapter 1 – Introduction.

Localised 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 the season (Giles et al. 2006), culture species (Jaramillo et al. 1992) and environmental conditions (e.g. tidal currents, water depth, riverine inputs).

Like mussels, oysters are filter feeders and farms produce waste products in the form of faeces and pseudofaeces and, accordingly, can cause similar types of enrichment effects. 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 (Kusuki 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.

Direct biodeposition effects associated with oyster cultivation are highly localised in 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 with that described for subtidal mussel culture in New Zealand (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). Effects associated with smothering are less likely to be important due to the relatively low densities per square metre of seabed associated with intertidal farming. Extreme enrichment effects in relation to oyster farming have been described only for suspended culture systems in Japan, and where they were attributed to repeated culturing and over-stocking (Ito & Imai 1955; Kusuki 1981).

A significant quantity of live and dead mussel and oyster material can also accumulate on the seafloor, produced

Table 3.11: Organic enrichment due to biodeposition from filter-feeder aquaculture – widespread effects.

Description of effect(s)	Biodeposition from bivalves is insufficient to directly influence far-field enrichment. However, intensive farming of filter feeders has the potential to reduce natural deposition rates by reducing the standing biomass of phytoplankton. The scope for dispersal of biodeposits from oysters is even more limited due to the depth and density of shellfish per area of seabed.
Spatial scale	Potentially *regional* (one to tens of kilometres from the farm).
Duration	*Short* to *medium term* (low level enrichment).
Management options	Unnecessary.
Knowledge gaps	The relationship between farming intensity and phytoplankton biomass depletion potential. Links between natural (background) sedimentation rates and propensity for organic enrichment from increased biodeposition.

* Italicised text in this table is defined in chapter 1 – Introduction.

primarily during harvesting and farm maintenance (see Section 3.3.2.5 for more information). This shell debris may promote the accumulation of fine sediment and organic matter by dampening currents and reducing oxygen percolation into the sediment, and in doing so, reducing the rate of mineralisation of organic matter (D. Morrisey pers. comm.). Excessive deposition and decay of fouling biomass may also exacerbate the organic enrichment described above, although such effects would be likely to be patchy beneath cultivation areas.

Deposition of fouling biota may also contribute to seabed enrichment beneath mussel and oyster 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. In such situations, the deposited fouling biomass may exacerbate enrichment effects (at least in the short term) associated with other processes (e.g. biodeposition).

Widespread biodeposition

Biodeposition from bivalves is considered less likely to result in far-field enrichment than finfish farming due to the comparatively low rates of biodeposit production. This is particularly the case for high energy sites, where resuspension and dispersion processes are such that the benthic impacts can be difficult to detect within 200 metres of a mussel farm (Hartstein & Stevens 2005). Conversely, intensive farming of filter feeders has the potential to reduce natural deposition rates by reducing standing biomass of phytoplankton and other suspended material. Reduced natural deposition may lead to an impoverished infauna and reduce benthic production.

3.3.2.2 Changes to physico-chemical properties of the sediments

Numerous studies overseas and in New Zealand have documented changes to the physico-chemical 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, sulphate reduction (Dahlbäck & Gunnarsson 1981) and lowered redox potential (Christensen et al. 2003; Grant et al. 2005).

Elevated sediment organic content is commonly encountered beneath mussel farm sites in New Zealand. Hartstein and Rowden (2004) found levels double that of reference location sediments at two sheltered mussel farm sites in the Marlborough Sounds. However, levels beneath a high energy site were similar to those observed in reference locations, highlighting how a dispersive environment can help reduce the intensity of seabed effects. Data from multiple assessments conducted under conventional Marlborough Sounds mussel farms indicated that, on average, sediments had only slightly elevated levels of organic material (about a 7.5 percent increase – based on a 1.5 percent increase in AFDW). In most cases, this level of organic enrichment increases the productivity of coastal sediments without major disruption to community composition.

3.3.2.3 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-scopic 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 affecting denitrification rates. Similarly, meiofaunal (very small organisms 0.45–1.0 mm long) community composition can change significantly due to the presence of elevated organic content beneath mussel farm sites (Mitro et al. 2000).

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), with consequently reduced mineralisation of organic waste.

In terms of infaunal community composition, grey literature from numerous studies conducted within the Marlborough Sounds and Firth of Thames indicates that animal abundance tends to be slightly elevated directly beneath mussel farms. However, as for finfish farms, enrichment is variable amongst sites, depending on environmental conditions such as depth and average current velocity and, as a result, species richness can either be either slightly depressed or slightly enhanced. Generally, the level of compositional change is that of a mild, positive enrichment effect rather than a major disruption to the functional integrity of the sediments. The abundances of opportunistic polychaetes tend to be slightly elevated, while the composition of other major infaunal groups (e.g. molluscs, crustaceans, echinoderms) remain comparable between farmed and unfarmed locations; that is consistent with an enrichment stage of about three (Figure 3.3). Higher level enrichment effects (above ES 3) are likely to be observed at farm sites that are predisposed to impacts (i.e. shallow, low flow).

3.3.2.4 Effects on epibiota

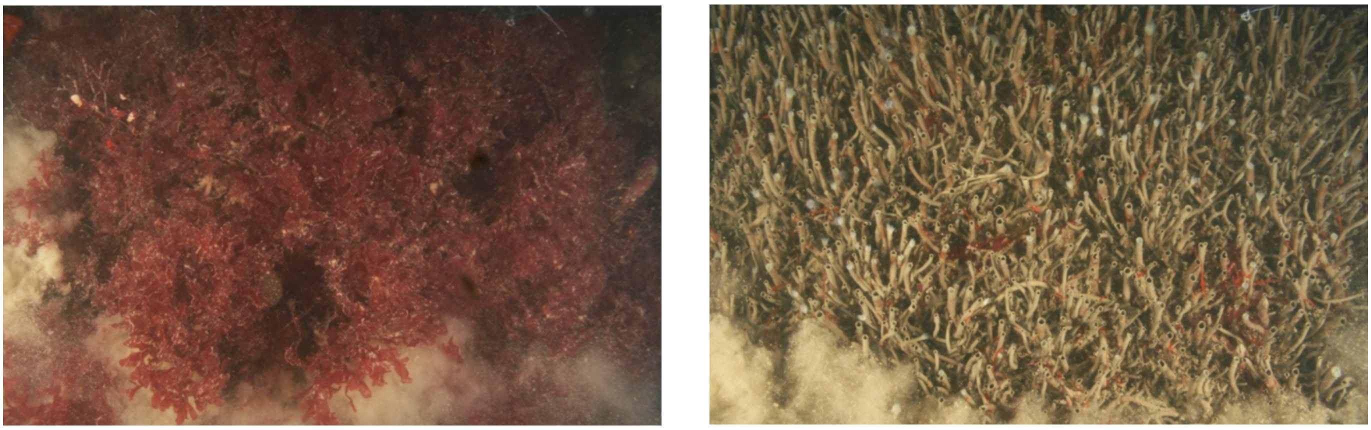
Depositional effects from mussel farms in New Zealand on plants and animals living on the surface of the seabed (referred to as "epibiota") are not well documented. The few studies that do exist describe reef-type communities (Kaspar et al. 1985; De Jong 1994) and an increase in predators (Kaspar et al. 1985; Grant et al. 1995; Inglis & Gust 2003) associated with shell drop-off below marine farming structures (described in Section 3.3.3). But there is very little information relating to 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 differences in epibiota distribution to the effect of mussel farming based on statistical comparisons.

In the absence of relevant published literature, Keeley et al. (2009) summarised observations from an investigation into tubeworm and red algae densities beneath mussel farm sites in Port Underwood, Marlborough Sounds (Figure 3.5). In this case, polychaete tubeworms and red algae deemed to be of “special ecological value” (DoC 1995) were observed on the seabed beneath and adjacent to proposed marine farm sites in Port Underwood. Follow-up investigations found much higher densities of tubeworms (average cover about 25 percent) outside the boundaries of existing farms compared with beneath existing farms (average cover less than 5 percent). 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 (average cover about 40 percent) than beneath existing marine farms (average cover less than 5 percent).

The National Institute of Water and Atmospheric Research (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 and so on) using baseline data collected prior to farm development. During the two years of farm development, LAC were exceeded along some transects for variables such as the 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 ecological importance (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).

Figure 3.5: Photographs of red macroalgae and tubeworms observed on the seabed in Port Underwood, Marlborough Sounds



3.3.2.5 Crop and biofouling drop-off and debris

Table 3.12: Habitat creation and biofouling due to filter-feeder aquaculture.

Description of effect(s)	Elevated or suspended structures provide novel habitat for fouling organisms and reef-type biota. Diversity and productivity can be locally enhanced, but may also facilitate the expansion and/or proliferation of invasive (unwanted) species. In the case of oysters, biofouling is primarily by intertidal species.
Spatial scale	*Local* to *regional* – Primary effects very localised (i.e. on structures), but dispersal via planktonic propagules can affect wider area and has potential implications for wider ecosystem.
Duration	*Short* to *long term* – Local colonies removable with removal of structures, but wider ecological invasions could be permanent.
Management options	Available habitat is partially controllable through the types (shape, composition of materials) and amount of structures that are used. The use of antifoulants on non-crop holding materials will also minimise fouling, as will controlling potential transfer vectors.
Knowledge gaps	None identified.

* Italicised text in this table is defined in chapter 1 – Introduction.

Summary

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; author's 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 sea stars, crabs and fish. In other situations, mussel clumps and shell litter can remain relatively barren of reef-type communities (Watson 1996, Figure 3.6).

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 number 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 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 green-lipped mussel farms in New Zealand, where densities can be 39 times higher than at non-farmed sites (Inglis & Gust 2003; author's 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 at existing mussel farming sites in New Zealand or overseas.

The accumulation of live oysters, oyster 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. These surfaces potentially provide novel habitats for fouling organisms and associated mobile biota, which would otherwise not occur (or would be present 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 and birds) that may be absent or at reduced densities in adjacent unvegetated soft-sediment habitats (Ruesink et al. 2005 and references therein).

The extent of oyster drop-off to the seabed is dependent 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 extent to which structures become fouled and also patterns of natural drop-off, or active defouling, by farm personnel. Subsequent effects on benthic community composition, for example, aggregation of carnivorous and deposit-feeding species in response to the food supply (e.g. sea stars, sea cucumbers) and competition between deposited shellfish and benthic filter feeders, are indicated for other forms of bivalve aquaculture (Smith & Shackley 2004; Hartstein & Rowden 2004) and conceivably occur in the case of intertidal oyster culture.

An important environment consideration in the case of oyster farms is 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 3.6: Shell debris beneath a mussel farm site in Kauauroa Bay (Marlborough Sounds), showing a sea cucumber and starfish foraging across shell litter (top left and bottom) and macroalgae growing on live mussels deposited on the seabed (top right)

3.3.2.6 Seabed shading by structures

Table 3.15: Shading of seabed by structures on filter-feeder aquaculture operations.

Description of effect(s)	Shading from farm structures could reduce the amount of light to the seafloor, thereby reducing the productivity of ecologically important primary producers such as benthic microalgae, beds of macroalgae or seagrass, with a range of associated ecological effects.
Spatial scale	*Local* scale – Very localised.
Duration	*Short* to *medium term* – Reflecting response rates of micro-algae and macro algae.
Management options	Limit farming intensity/spacing of lines.
Knowledge gaps	Information relating to both and/or percent reductions in Photosynthetically Available Radiation (PAR) in relation to farm arrangements and benthic ecological sensitivity to PAR reductions.

* Italicised text in this table is defined in chapter 1 – Introduction.

Summary

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 the Kaipara Harbour.

For mussel farms, 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 were abundant directly beneath the farm structures. Shading effects are conceivably of most importance where 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. However, in New Zealand at least, mussel and oyster farms are usually intentionally situated to avoid such habitats. Shading effects are likely to be site specific and can be effectively mitigated by appropriate farm site selection.

3.3.3 Factors relating to all benthic impacts

3.3.3.1 Main factors affecting the extent of seabed effects

The magnitude and spatial extent of seabed effects from mussel and oysters farms (Sections 3.3.2.1 to 3.3.2.6) are a function of a number of inter-related factors that can be broadly considered as farm attributes and physical environment attributes.

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, Hartstein & Rowden 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).

Similarly, the magnitude of effects from oyster biodeposition will depend primarily on 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).

Farm attributes

Culturing and husbandry techniques have been identified as having the potential to influence the magnitude of seabed effects beneath shellfish farms; 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.

In New Zealand, Pacific oysters are traditionally cultured in the intertidal zone on racks or in baskets but they also appear suited to subtidal culture, which opens up a variety of different methods. The relative environmental effects of culturing different bivalve species in suspension were considered by Gibbs et al. (2006) using available, pertinent physiology literature and likely culture techniques. The key result was that mussels generally appear to exhibit the highest clearance and excretion rates of the bivalves considered.

Similarly, biodeposition intensity greater than 400 grams per day per 1000 individuals occurred most frequently in mussels (40 percent), followed by scallops (33 percent), cupped oysters (29 percent), flat oysters (11 percent) and, finally, clams/cockles (6 percent). Hence, the propensity for Pacific oysters to induce benthic or water column effects is expected to be comparable with or less than that of green-lipped mussels. Therefore, seabed effects arising from subtidal oyster cultivation are likely to be analogous to those described for subtidal mussels.

Physical site attributes

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 (Hartstein & Rowden 2004), it also increases oxygen delivery to the sediments, allowing for more efficient breakdown (i.e. mineralisation) of organic material (Findlay & Watling 1997). Therefore, as for finfish farms, deep sites (over 30 metres) located in areas of strong water currents will have depositional footprints that are less intense and more widely dispersed than shallow, poorly flushed sites. Conversely, where currents are very weak or water depth is shallow, biodeposition would be expected to result in moderately enriched sediments.

Changes in seabed topography

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 & 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 than to increased sedimentation rates. In New Zealand, sediment build-up to the top of Pacific oyster racks 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 unuseable and farming abandoned, with shell litter and debris still evident many years later. The redistribution of sediments either into (Kirby 1994) or out of (Mallet et al. 2009) culture sites may also occur in relation to events such as storms that lead to large-scale sediment mobilisation.

3.3.3.2 Recovery

Recovery rates of seabed communities from deposition-related enrichment effects of mussel and oyster farms have not been well described but are assumed to be site specific and relatively rapid once farming ceases. Based on literature for a mussel farm (Mattsson & Lindén 1983) and several more highly impacted fish farms (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 to strong enrichment has occurred. Accumulated shell material from drop-off is likely to persist in the sediment beyond the point of recovery from typical enrichment by type effects. Sticks and other inorganic debris, more commonly associated with 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, culture

Table 3.14: Changes in seabed topography due to filter-feeder operations.

Description of effect(s)	Seabed typography beneath oyster farms can be changed (by tens of centremetres) from erosion or accretion of sediments around the structures.
Spatial scale	*Local* scale – Very localised – around and directly beneath structures.
Duration	Unknown. *Medium* to *long term*. May be site specific (degree of alteration on strength of currents at site.
Management options	Optimising alignment of rows and types of structures used.
Knowledge gaps	None identified.

* Italicised text in this table is defined in chapter 1 – Introduction.

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.

3.3.4 Impact mitigation and management strategies

3.3.4.1 Environmental variables and quality standards

As with fish farms, the primary benthic effects that occur beneath mussel and oyster farms concern enrichment and are, accordingly, assessed with a similar suite of environmental variables. That is, infauna taxonomy and the associated biotic indices are the primary indicators of benthic condition, in conjunction with some, or all, of the usual suite of physico-chemical indicators. These include observations of sediment colour, odour, redox potential discontinuity layer, sulphide concentrations and sediment organic content (Wildish et al. 1999, Keeley et al. 2009). Of these, sediment organic content has proven particularly useful and is often included (along with other indicators) in marine farm monitoring programmes in New Zealand and overseas.

The levels of acceptable impact, or EQS, associated with shellfish farm assessments are, however, different in accordance with the expectation of milder levels of enrichment. Using the enrichment scale that is described in Table 3.7 and depicted in Figure 3.3, the benthic conditions associated with mussel farms would not be expected to exceed approximately ES 4 (although the scale is yet to be formally, quantitatively validated for finfish farms). While this may appear to be a double standard, applications for mussel farms are made on the basis that they will result in only moderate levels of enrichment. Therefore, the size, number and positioning of established mussel farms are designed to only result in mild levels of organic enrichment.

3.4 Lower trophic level species

3.4.1 Overview of seabed effects

The two potential lower trophic level culture species assessed here are the sea cucumber, *Australostichopus mollis*, and the macroalgae, *Undaria pinnatifida*. Other potential culture species exists and a summary of what is known about the potential ecological effects of farming those species can be found in Keeley et al. 2009. While sea cucumbers and *Undaria* have been grouped together here as "lower trophic level" species, they are very different organisms and, as such, are discussed separately.

3.4.2 Sea cucumbers

Seabed effects resulting from culturing sea cucumbers are difficult to assess because they are yet to be commercially cultured in New Zealand and the methods that may be used and the size and intensity of the operations remain undetermined. There are a variety of potential methods, such as land based, seabed ranching, multi-trophic level arrangements (beneath mussel farms) and reseeding of the natural population for subsequent harvesting (pers. Comm. K Heasman, Cawthron Institute). Some of these methods (e.g. suspended culture) will require additional feed inputs, while others may be supported by the by-products (biodeposits) produced by other forms of aquaculture (e.g. mussel farming) and thereby effectively mitigate their enrichment effects.

Sea cucumbers are deposit feeders and obtain 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). 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 culturing other species. Hence, they are becoming a popular co-culture candidate species with bivalve (e.g. oysters, Paltzat et al. 2008), paua (Kang et al. 2003) and fish farms (Ahlgren 1998). The ecological issues associated with culturing *A. mollis* remain undescribed and are unlikely to be realised until the species is cultured in significant quantities. Nevertheless, it is reasonable to assume that the potential for seabed enrichment-type effects is less than has been described for finfish, and to a lesser degree, shellfish aquaculture.

Table 3.15: Organic enrichment from biodeposition from lower trophic level aquaculture.

Description of effect(s)	Unwanted sediments are excreted once the organic components are digested. The potential for seabed enrichment-type effects is less than has been described for finfish or shellfish aquaculture and may in fact mitigate enrichment effects of those activities.
Spatial scale	*Local* scale.
Duration	Reversible within the *short term* to *medium term*.
Management options	Site selection, reducing stocking densities.
Knowledge gaps	None identified.

* Italicised text in this table is defined in chapter 1 – Introduction.

3.4.3 *Undaria*

The only macroalgal 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 Aquaculture Management Area (AMA) on Banks Peninsula. The volumes of *M. pyrifera* being harvested are small and it colonises the structures naturally, so the method can only be loosely described as aquaculture. *U. pinnatifida* is also presently a "by-product" of the mussel industry as it grows profusely on the upper parts of mussel lines.

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 to mop up excess nutrients discharged from fish farms (Zhou et al. 2006; Blouin et al. 2007; Kang et al. 2008; Xu et al. 2008). While macroalgal 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 algal farms, which can impede growth in adjacent seagrass beds and alter the macrofauna community contained within (Eklöf et al. 2005). Another possible seabed effect relates to the sloughing off, or storm-induced removal of, algae from a farm, which may then accumulate on nearby coastal margins where it may decompose and smother the benthos. However, drift macroalgae also comprises an important part of the beach ecosystem, providing habitat and food for some coastal invertebrates (Marsden 1991). Otherwise, seabed effects resulting from algae culture are expected to be relatively minor or even negligible.

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Appendix 3.1: Review of Environmental Quality Standards (EQS) used internationally.

	Feed added	Filter feeders		Other species
Effect		Mussels	Oysters	
Organic enrichment	1–24	1–5, 8, 9, 13, 14–24	1–5, 8, 9, 13, 14–24	
Smothering	1–24	1–5, 8, 9, 13, 14–24	1–5, 8, 9, 13, 14–24	
Habitat creation and biofouling	?	?	?	
Drop-off and debris	1–5, 11, 13	1–5, 11, 13	1–5, 11, 13	
Changes in topography	NA	NA	Visual	
Shading	? 1–24?	?	?	
Wider effects – deposition	1–5, 14–20	NA	NA?	
Wider effects – dissolved nutrients on epibiota	21–24, + targeted reef monitoring	NA	NA?	
Wider effects – phytosanitary composition on epibiota	NA	21–24, + targeted reef monitoring	NA?	

Note: Numbers 1–24 refer to the environmental variables (as listed in Appendix 3.2) that are used to assess the different types of effects.

Appendix 3.2: Types of environmental variables and associated Environmental Quality Standards (EQS) used in conjunction with salmon farming in New Zealand and internationally

	Group		Variable	Where used	EQS use	Standards
Benthic	Physical	1	Sediment grain size	NZ, Ca, Ch, No, UK, US	–	
	Chemical	2	Redox	NZ, Ta, Ca, I, Ch, No, UK, US	US, UK, Ta, WWF	Various – spatially explicit
		3	Sulphides	NZ, Ta, Ca, Ch, No, UK, US	US, UK, Ta, WWF	Various – spatially explicit
		4	pH	Ch, No	–	
		5	Trace metals (Cu and Zn)	NZ, Ta, US	UK, Ta, NZ	ANZECC ISQG-Low-High
	Observations	6	Feed pellets	NZ, Ta, Ca, I, Ch, UK, US	Ta, I, UK	Qual. categories. High presence prohibited
	(Qualitative)	7	Out-gassing	NZ, Ta, I, Ca, Ch, No, US	Ta, NZ	Free out-gassing prohibited
		8	Odour	NZ, Ta, I, No, UK, US	–	
		9	Colour	I, No, NZ, Ta, UK, US	–	
		10	Sludge thickness	I, No, UK, US	–	
		11	Consistency	Ca, Ch, I, No	–	
		12	Beggiatoa mat	NZ, Ta, Ca, I, Ch, UK, US	US, UK, I, Ta, NZ	Qual. categories. High coverage prohibited
		13	Rubbish/debris	?	–	
Biological	Infauna	14	Abundance	NZ, Ta, Ca, I, Ch, No, UK, US	US, UK, Ta, NZ	Azoic prohibited, limits on abundance relative to reference
		15	No. Taxa	NZ, Ta, Ca, I, Ch, No, UK, US	US, UK, I, Ta, NZ, WWF	Limits on minimum number of taxa
		16	No. opportunists	?	US, Ta	Limits on total abundance of
		17	Shannon Diversity	?	WWF	Minimum diversity restrictions
		18	Evenness	?		
		19	AMBI	?	WWF	Minimum diversity restrictions
		20	Other indices (ITI, BQI)	?	WWF	Minimum diversity restrictions
	Epifauna		Non-specific use	Ca, I, NZ, Tas, US		
		21	Presence/absence			
		22	Diversity			
		23	Observed health			
		24	Aggregations			

pH = acidity; Cu = copper; Zn = zinc; NZ = New Zealand; Ta = Tasmania; Ca = Canada; Ch = Chile; N = Norway; I = Ireland; UK = United Kingdom; US = United States of America; WWF = World Wildlife Fund Draft Aquaculture standards (http://www.worldwildlife.org/what/globalmarkets/aquaculture/WWFBinaryitem21275.pdf). Summarised from Wilson et al. (2009) and various online sources.