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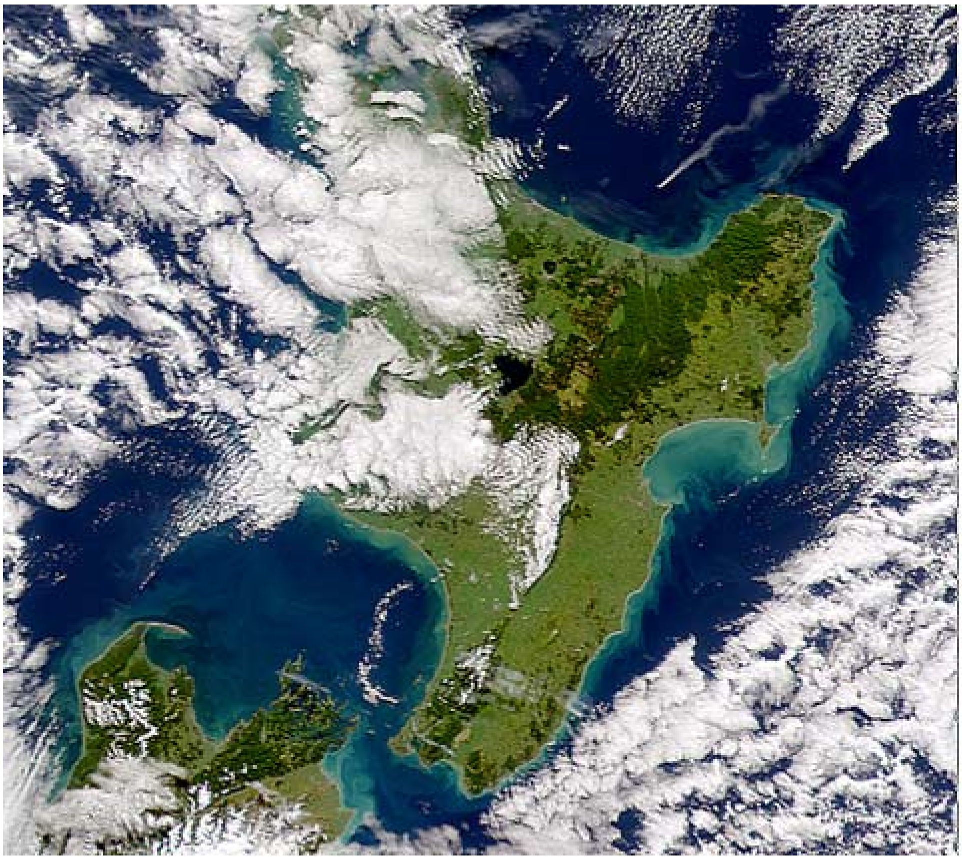

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EXECUTIVE SUMMARY

Morrison, M.A.; Lowe, M.L.; Parsons, D.M.; Usmar, N.R.; McLeod, I.M. (2009). A review of land-based effects on coastal fisheries and supporting biodiversity in New Zealand.

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Land-based effects on coastal fisheries may occur through a diversity of mechanisms. Changing inputs from the land have included large volumes of suspended sediments and nutrients into the coastal zone, following large-scale clearances of New Zealand's forests, and the expansion of land-based industries such as pastoral livestock farming, dairying, and exotic plantation forestry. More localised effects from urbanisation have included elevations of heavy metal concentrations and pollution from sewage. Impacts from such activities have continued into the present day. Commercial coastal fisheries have been established over the same time period, with initial periods of heavy utilisation leading to over-fishing of many stocks, and subsequent catch reductions to more sustainable levels. Most fisheries are now managed under the Quota Management System which generally applies Maximum Sustainable Yield (MSY) targets, under which stocks are fished down to a level where productivity is thought to be highest. This inevitably leads to large reductions in overall (meta-) population size, and fishing down of the larger and older size/age classes. One assumption inherent in most stock assessment models used to provide advice on changes to catch limits (and of the generally stable catch limits for stocks for which there is no stock assessment model) is that fished populations will move back towards their original size following any reduction in fishing pressure. It is also assumed that environmental influences on the stock and the carrying capacity of the system remain constant over time or fluctuate without much trend. However, substantial changes in estuarine and coastal habitats and ecosystems are known to have occurred over the last 100 or more years, and to still be occurring. These environmental impacts have happened over the same time frame as that of the establishment, and subsequent over-fishing of, coastal fisheries, and have driven population trends in the same direction, i.e., in a negative direction, for most species (but not all). Such impacts are currently poorly understood, with most fisheries research having been directed at the fished species themselves, in terms of factors such as how many there are, growth rates, age structures, and fishing removals, and the integration of these variables into numerical single species population models. The possible effects of environmental and habitat degradation on these fished populations have been largely ignored.

In New Zealand, arguably the most important land-based stressor is sedimentation, including both suspended sediment and deposition effects, and associated decreases in water clarity (which may also be driven by nutrient effects). Impacts may be direct on the species themselves, such as clogging of the gills of filter feeders and decreases in filtering efficiencies with increasing suspended sediment loads (e.g., cockles, pipi, scallops), reductions in settlement success and survival of larval and juvenile phases (e.g., paua, kina), and reductions in the foraging abilities of finfish (e.g., juvenile snapper). Indirect effects include the modification or loss of important nursery habitats, especially those composed of habitat-forming (biogenic) species (e.g. green-lipped and horse mussel beds, seagrass meadows, bryozoan and tubeworm mounds, sponge gardens, kelps/seaweeds, and a range of other 'structurally complex' species). For instance, while we still have much to learn, recent work using otolith chemistry strongly suggests that west coast North Island snapper populations (SNA 8), from Cape Reinga to Wellington, largely originated as juveniles from the Kaipara Harbour. Within this harbour, juvenile snapper are found in association with nursery habitats composed of horse mussel beds and seagrass (especially subtidal) meadows. These habitats are known to have been impacted by historical land-use practices and continue to be under pressure, especially from sedimentation from the surrounding catchment. This means that the carrying capacity (for snapper) of the system that supports the SNA 8 fishery may have declined substantially over the past 100 years. In addition, the coastal stock has been fished down to a low biomass, with most of the old and large fish being removed, so

that the fishery is now reliant on just a few year classes. This has reduced the resilience of the stock, so that several sequential years of low recruitment could result in the fishery becoming uneconomic due to very low population densities. Before this, a large reserve of many age classes would have ‘buffered’ the stock for several decades. Thus, the SNA 8 stock is under at least two types of stress. Similar issues are likely to exist for other harvested species that have nursery grounds close to shore.

International work has shown that eutrophication has the potential to *initially* increase primary productivity (phytoplankton and macrophytes), and then to create profound cascades of effects into marine ecosystems, including loss of seagrasses, and eventually macrophytes, increases in phytoplankton blooms that reduce light levels reaching the sea-floor, and subsequent oxygen depletions as blooms die and increase detrital levels on the seafloor, and large-scale losses of benthic prey assemblages that support finfish fisheries. Factors that moderate the influence of these processes include tidal streams, the degree of water transport across different areas, and the presence of large numbers of filter-feeding bivalves. Loss of such bivalve populations, e.g., from over-harvesting or sediment impacts, may exacerbate other land-based stressors, such as eutrophication, through reducing the resilience of local systems. Little work has yet been done on the potential impact of eutrophication on coastal fisheries in New Zealand, though it may be modest relative to other areas of the world. Other pollutants are generally associated with urbanisation, and as such are generally more localised in extent, and at relatively low, though sometimes ecologically influential, concentrations compared to other industrialised countries.

We suggest that there are substantial gaps in our knowledge of how land-based stressors affect coastal fisheries both in New Zealand and globally, in particular through mechanisms of sedimentation in the New Zealand context. These stressors, and their impacts, cannot be considered in isolation from other stressors, such as fishing, which are likely to interact synergistically on harvested species populations. Suggestions for research on these stressors, designed to help uncover and address impacts important for both land and fisheries managers to address, include: fundamental and systematic inventorying of fisheries species/habitat associations for different life stages, including how changing habitat landscapes may change the relative production of different fished species; better knowledge of connectivity between habitats and systems at large spatial scales, where impacts at one location may have far-field cascades into distant areas through subsequent fish movements; the role of river plumes in affecting local ecosystem processes; the effects of land-based stressors both directly on fished species, and indirectly through impacts on nursery habitats including plants (e.g., seagrass meadows, kelp forests, maerl beds) and animals (e.g., mussel beds, bryozoan and tubeworm mounds, sponge gardens); a better spatially based understanding of the integrated impacts of land-based and marine-based stressors on coastal marine ecosystems; and associated spatial mapping and synthesis to provide both decision support management systems, and as research tools that can help direct and interpret new research initiatives. With climate change predicted to increase both the frequency and intensity of storms and rainfall events, and intensification of land use, the relevance of addressing such issues is likely to increase.

1. INTRODUCTION

New Zealand's coastal environment and associated habitats support valuable invertebrate and finfish fisheries. Most of these fisheries are now fully exploited. Many have a history of heavy exploitation in their initial phases, which has subsequently proven to be unsustainable, and contemporary catches are now at lower levels than in the past. For most fished species in the coastal zone (and beyond), we have little knowledge and understanding of their habitat requirements over their life cycle, with a few notable exceptions on shallow rocky reefs (e.g., paua, kina, rock lobster), and on intertidal soft shores (e.g., cockles, pipi). Even for these species, many significant knowledge gaps remain. Most research on fished species has focused on measuring aspects of their population dynamics over time, including abundance (biomass), size and age structures, and growth, and the integration of these variables into single species numerical models of population dynamics. As most fished species are now within the Quota Management System (QMS), there has also been a central focus on managing towards Maximum Sustainable Yield (MSY). This approach assumes that there is a level of biomass at which the productivity of the stock is maximised, and management is aimed at this level (which is invariably substantially lower than the original biomass of the population before fishing commenced). This approach also results in most of the larger and older individuals being removed from the population, as the stock is 'fished down'.

The MSY approach assumes that the carrying capacity of the ecosystem is relatively constant, so that if the fishing effect is reduced, then the stock will increase back towards its pre-fishing state. This viewpoint implicitly assumes that fishing has been the only stressor on fished populations (and the wider ecosystem), and that once it is removed, populations will recover. However, over the last century or more, almost all anthropogenic (human-induced) impacts and associated signals have trended in a similar direction, i.e., towards increased stress on natural freshwater and inshore systems, as well as on semi-enclosed marine ecosystems (Caddy 2000). We define a stressor here as "*a variable that, as a result of human activity, exceeds its range of normal variation (Auerbach 1981), and adversely affects individual taxa or community composition*" (Townsend et al. 2008). This means that in addition to fishing, many other stressors have been operating on marine systems over the same time scale, including sedimentation, eutrophication, and pollution. New Zealand, while comparatively recently intensively settled by humans, has not escaped these effects. Given our intensive use of the land to support our production-based economy (i.e., sheep and cattle, dairying, forestry, viticulture, and cropping), as well as the development of many coastal towns and cities including port infrastructure, land-based activities have resulted in significant impacts on our adjacent coastal ecosystems, and by extension are also likely to have had significant cascades into the fisheries that they support.

These impacts are quite sparsely researched and understood, and addressing them has been largely absent from fisheries management until very recently. The indirect impacts of actual fishing (e.g., habitat destruction, and removal of keystone species and ecosystem engineers) also fall into this category, and are synergistic with land-based impacts. Habitat degradation and loss often occurs slowly and incrementally over long time scales that may exceed that of a human lifetime. This means that each subsequent human generation has a quite different view of what is pristine and natural in the oceans, referred to as "shifting baseline syndrome" (Dayton et al. 1998, Jackson 2001), and so the magnitude of change is usually seriously underestimated. Such habitat and ecosystem impacts are only now beginning to be meaningfully acknowledged by humans. For example, in Europe less than 15% of the coastline is considered to remain in good condition, with near elimination of many productive and diverse coastal habitats (Airoldi & Beck 2007). Similarly, a comparison of 12 estuarine and coastal ecosystems in North America, Europe, and Australia by Lotze et al. (2006) found human impacts to have depleted 90% of formerly important species (including many habitat-builders), destroyed 65% of seagrass and wetland habitat, reduced water quality, and accelerated species invasions. Impacts on many of these habitats, especially subtidal ones, are very poorly documented, and in many cases may never be fully known. At present, there seems to be limited public, political, and even scientific awareness of the extent, importance, and consequences of such a long history of coastal habitat loss (Lotze 2004).

In this short review, we assess the current state of knowledge of the impacts of land-based activities on coastal fisheries and their supporting habitats in the New Zealand context, using case studies where they exist, augmented by overseas work and examples.

1.1 Objectives

In March 2008 the Ministry of Fisheries asked NIWA to write a short review, aimed at scientists and resource managers, covering the following broad discussion areas:

- A very brief introduction to coastal fisheries in New Zealand: recreational, customary, commercial.
- A summary of what land-based effects are thought to be important (sedimentation, eutrophication, organic pollution, heavy metals, etc).
- A description of the likely mechanisms of impact (e.g., clogging of filter-feeders, light declines for plants, loss of structured habitats for juveniles), and relative scale of impact on fisheries and biodiversity (e.g., recruitment, stock numbers).
- An analysis of what fisheries are (or are likely to be) affected, including invertebrates and fish, and different habitats (soft sediment, rocky reef, inshore pelagic; intertidal, estuarine, coastal, islands).
- Some indication of the relative magnitude and significance of effects on different fisheries.
- An indication of key locations around New Zealand where land-based effects are likely to have significant impact on fisheries productivity or biodiversity.
- Selected short case studies where sufficient information exists. Species-focused case studies would probably include snapper, cockle/pipi, toheroa, kina/paua, and scallops. Biodiversity examples might include broadly defined biogenic habitats (including mangroves, seagrass, horse mussels, sponges), and interactions between these and fisheries.
- A concise summary of historical and current relevant research projects and some gap analysis and suggestions.
- Links, information sources, resources, contacts, etc.

1.2 Scope and limitations of review

Written material, both from the primary and grey literature, was sourced using a combination of web search engines (Scopus, Google), manual searches of New Zealand science journals, and professional contacts in both science and regulatory agencies. The quality and type of material varied widely across different sources. Grey literature was included as this held important information in the New Zealand context that did not exist in other forms. Given the limited scope of the review, we deliberately and explicitly focussed on either species that directly supported fisheries, and/or species that played a pivotal role (known or suspected) in directly underpinning fisheries production, i.e., habitat formers. This also held for processes and effects – while we refer to the wider ecosystem where relevant, our main focus was on coastal fisheries. For example, while sedimentation strongly affects soft sediment benthic assemblages in general, we specifically focussed on fisheries species such as cockles, pipis, and scallops; and habitat forming species such as horse mussels, sponges, and bryozoan mounds. However, we fully acknowledge that, in reality, fished species are integrated components of the overall ecosystem, and their responses to land-based impacts do not occur in isolation from the ecosystems in which they have evolved and live.

We have been systematic in our use of New Zealand material, but given its patchy nature (including many significant information gaps) have augmented it wherever possible with relevant international knowledge, to establish an overall framework (while mindful of the limited time resources available for this review). We hope that this report will act as a catalyst for new research and management initiatives to address land-based impacts on coastal fisheries and their supporting ecosystems, and that a more habitat-based ecosystem management of coastal fisheries will eventually emerge.

2. THE NEW ZEALAND SITUATION – PHYSICAL GEOGRAPHY

2.1 The pre-human past

The New Zealand land-mass (Figure 1) has a diverse range of geology and land types, a number of which are highly erodible. Combined with a history of active tectonics, strong storm and rainfall dynamics, and in recent human times, changing land use, this has resulted in significant inputs of terrestrially derived material to estuaries, the coastal zone, and beyond. Before humans arrived, New Zealand was substantially covered in various forest types and scrubland. Records from lake basins, which record forest disturbance events as stratified layers of deposited sediment, give some information on what natural dynamics were once like. Examination of sediment cores from two lakes in the Hawkes Bay (Tutira and Putere Districts) (Wilmshurst 1997, Wilmshurst et al. 1997), found that before humans arrived, the composition of the adjacent forests fluctuated frequently due to disturbance from fires generated by lightning, droughts, and a major volcanic eruption. Each natural disturbance event (as shown by short-term increases in succession species pollen and other material) was followed by the full re-establishment of the forest. Storms (cyclones) were not a major disturbance to lowland podocarp/hardwood forests. The main effect of storms was to generate a sudden increase in the rate of surface run-off and fluvial transport, causing scouring and rapid transportation of riverbank sediments into the lakes. Although soil erosion and slipping may have occurred on a small scale in the catchments, standing vegetation and debris trapped such sediment and held it locally, preventing it from reaching the lake basins. This minimal effect of storms on vegetation and soil stability is in stark contrast to the damage storms now inflict through severe landslide erosion on unstable pasture-covered hill country in Hawkes Bay (Page et al. 1994). Clearance of this region started about *c* 800–500 years ago (Wilmshurst et al. 1997).

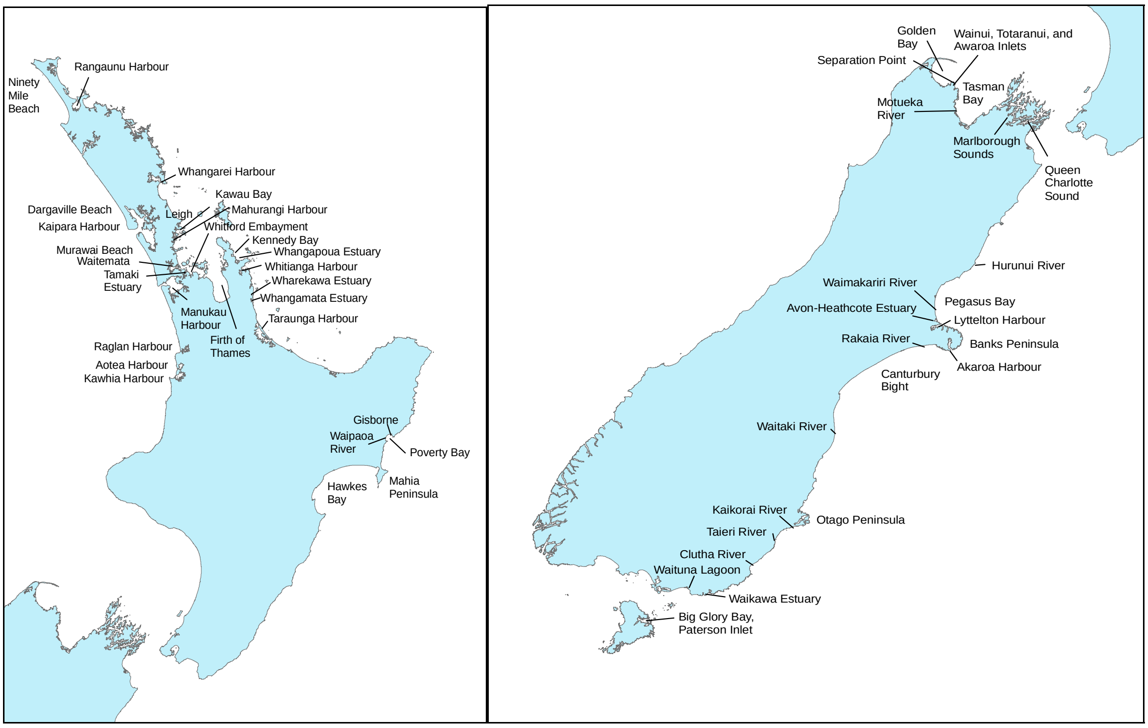


Figure 1: Locations of place names mentioned in the text, for the North and South Islands respectively

2.2 The arrival of humans

Following the arrival of Maori, around half of New Zealand's vegetation cover was removed by burning (Poole & Adams 1984). Most of the Maori land use was based around coastal plains or near lakes and rivers (Glade 2003), and hilly regions were only marginally influenced. Following the arrival of Europeans, humans moved into the back country, and converted extensive areas of hilly areas from native forest and bush into pasture (Figure 2). This reduced the strength of the regolith (the layer of loose, heterogeneous (mixed) material covering solid rock) and substantially increased the susceptibility of slopes to land slides (Glade 2003). Since that time, sediment production has been largely driven by landslide events. On unstable slopes, many thousands of landslides were triggered by high magnitude low-frequency climatic events during storms with return periods in excess of 50 years. In contrast, low magnitude, high frequency rainfall events have caused major gully and channel erosion (Glade 2003). Inputs of sediments to the coastal zone are now especially high by world standards, approaching almost 1% of total world sediment yields (Robertson & Stevens 2006).

Figure 2: Remnant of beach forest cut and burned in the 1920s–1930s, Huiarua Station, Tokomaru Bay. (Photo: P. Morrison (1974). Source: DOC.)

The amount of sediment yield into the estuarine and marine environment varies strongly depending on the presence of rivers, the amount of rainfall, and the erodibility of the catchment soils. For instance, the seafloor of the Hawkes Bay–Wairarapa region is also almost entirely covered by 'modern' sediments (clays and muds) resulting from the highly erodible sediments of the adjacent catchment, while a similar seafloor composition off the Nelson–Westland shelf is driven by high rainfall generating large sediment loads into rivers (Carter 1975). In strong contrast to these areas, the most northern and southern areas of New Zealand have seafloors typically covered by biogenic (calcium carbonate generated from the remains of living organisms such as shellfish and bryozoans), and/or relict (from earlier sea level shore-lines) terrigenous (land-derived) sediment (Carter 1975). Such seafloor habitats tend to also have high biodiversity values. In these regions the coastline is deeply embayed either by fiords or by drowned river valleys, and few rivers empty directly onto the shelf, with most flowing into coastal embayments (which act as sediment traps with their sheltered waters), and/or basin and sill bathymetry. Intermediate between these are the Waikato–Taranaki and Canterbury–Otago shelf areas, where sand dominates, with relict terrigenous and/or biogenic sediments dominating the middle and outer shelf. Factors influencing these patterns are that rivers in the Canterbury–Otago region tend to have lower sediment loads, while major rivers are largely absent in the Waikato–North Taranaki region (with a few notable exceptions, with associated local footprints of high mud content) (Carter 1975).

Where rivers empty out onto the shelf, sediment inputs can be high (Figure 3). Off the west coast of the South Island, rivers deliver about 12–26 million tonnes a year, resulting in a sedimentation rate on the shelf estimated to average 1–2 mm yr^{-1} (Probert & Swanson 1995). The annual sediment load yields of major rivers draining the western Southern Alps are 10 times higher than world average rates for mountainous areas (Griffiths 1979, see also Adams 1980). In the Wanganui Bight, sediment largely derived from the Waitotara, Wangauni, Rangitikei, and Manawatu Rivers contributes *c* 4.44 million tonnes yr^{-1} to the coast (Griffiths & Glasby 1985). With no tidal deltas, their mouths are dominated by wave dynamics, and relatively high mud percentages occur near shore, and increase with depth (Hayward et al. 1977). Associated with these rivers may be large sediment plumes, especially during and just after storm events.

Figure 3: Annual present-day suspended sediment yields into different coastal areas of New Zealand. Numbers given are tonnes x 10^6 per year; bracketed figures are predicted yields if rivers had no dams on them. (Source: Murray Hicks, NIWA.)

3. LAND-BASED ACTIVITIES' PHYSICAL CONNECTIONS TO THE MARINE ENVIRONMENT

3.1 A global issue

Estuarine and coastal ecosystems are now under significant pressure globally from human land-based activities. Some of the most serious problems include alteration and destruction of habitats and ecosystems, effects of sewage on human health, widespread and increased eutrophication, decline of fish stocks and other renewable resources, and change in sediment flow due to hydrological changes (GESAMP 2001). The global value of the goods and services provided by marine and coastal ecosystems is roughly double the value of those provided by terrestrial ecosystems, and is considered comparable with global GDP (GESAMP 2001). It is challenging to determine the relative importance of the different types of land-based activities that adversely affect fisheries productivity. Internationally recognised stressors include land-use and forestry practices which can result in increased sediment run-off, leading in turn to the loss of fish habitat through the smothering of seagrass beds and the siltation of coral reefs; physical destruction of reefs by tourism or mining; and poor water management practices that can have adverse effects on some estuarine fish and/or impede the spawning of anadromous fish (GESAMP 2001). However, until recently different stressors, both land-based and marine (e.g., fishing), have been considered in relative isolation by researchers and managers. In a review on coastal eutrophication, Cloern (2001) emphasised this problem, stating that "*Our view of the problem* [eutrophication] *is narrow because it continues to focus on one signal of change in the coastal zone, as through nutrient enrichment operates as an independent stressor; it does not reflect a broad ecosystem-scale view that considers nutrient enrichment in the context of all the other stressors that cause change in coastal ecosystems*". We strongly agree with this viewpoint. While the nature of this review is focussed on land-based impacts, we emphasise that these do not act in isolation from other stressors, such as the impacts on benthic habitats from fishing, and that populations stressed by one factor are generally more susceptible to additional stresses caused by other factors (Buchbaum et al. 2005). Following is a discussion on the different types of stressors acting in the New Zealand context, with inclusion of overseas material to help bridge large information gaps within the New Zealand context.

3.2 Sedimentation – physical processes

3.2.1 Wind and rain sediment generation, transport, and deposition

Three general processes are at work; *erosion*, where rock and soil particles are detached from the matrix they occur in; *transport*, where these materials are moved to a different place; and *sedimentation*, where these materials are deposited on the earth's surface again (Environment Canterbury 2007). Water and, to a much lesser degree, wind, are the main agents that drive these processes, mediated by the types of vegetation present. Raindrops can be travelling at 60 km hr^{-1} when they hit and detach soil particles by the transmission of kinetic energy and a hydraulic effect as they strike exposed soil surfaces ("*splash erosion*") (Environment Canterbury 2007). Flowing water scours away soil when the shear stress of the flow exceeds the ability of the soils to resist erosion, most noticeably around concentrated flows. Increasing water velocity and turbulence increases the rate of entrainment of soil particles into the transport process. As soils become more saturated with water, and the capacity of the existing surface dentition (water courses) is exceeded, excess water travels down-slope with gravity, carrying with it soil particles previously detached by raindrop impact and/or scour. These remain in transport until the energy level of the flow becomes too low to keep soil particles in suspension, and gravity deposits them on the bottom (Environment Canterbury 2007). Erosion by water can include: *sheet erosion*, where uniform thin layers of soil are removed by the force of shallow overland flows, which may cover large areas of sloping land; *rill erosion*, where tiny channels (rills) are removed by cultivation, and uniform sheet flows break up into more concentrated flow paths (more important with increasing slope length and/or gradient); and *gully erosion* (huge rills), where large and concentrated water flows form incised channels/gullies, that are very difficult and expensive to remediate (Environment Canterbury 2007). Wind erosion can also be important in some regions. For instance, the loess soils of Canterbury (Figure 4) are vulnerable to dry summers, with the combination of strong and dry northwest winds and lightly textured soils, may produce significant air pollution by fine particles. Processes of wind erosion include *creep*, where larger particles roll, slide, or are moved by the impacts of saltating particles (*saltation*; skipping or bouncing of particles along a surface). Where particles are small enough, they are lifted and carried away as dust (*suspension*) (Environment Canterbury 2007).

Figure 4: Oblique view of Banks Peninsula and Pegasus Bay (NASA satellite image, 3 April 2001), showing the turbid coastal zone with complex hydrodynamics, including loess erosion. (Source: Fenwick et al. (2003).)

Nationally, shallow landslides are the biggest source of sediments (Figure 5). These occur on all land types, and are caused by intense rainfall events. Landslide failure rates increase with slope, most commonly occurs on slopes of more than 20 degrees, with the highest overall contribution coming from slopes of 25–35° (Jones 2008). In absolute terms, a greater number of landslides occur in native forest, but this is strongly influenced by the fact that (remaining) native forest is often located at higher altitudes than plantation forests, on steeper slopes, and is subject to heavier and more frequent rainfall, all of which increase the probability of landslides. However, compared to pastoral farming, the presence of closed canopy forest significantly reduces the degree of erosion (especially landsliding) during large storm events (Jones 2008).

Figure 5: Soil-slip erosion on hill-country pasture. (Source: Ministry for the Environment.)

Vegetation cover is a very significant moderator of erosion potentials (Jones 2008). Work assessing the dynamics of landslides near Gisborne, in erosion prone hill country, before and during Cyclone Bola (1988), found that native and exotic forest more than eight years old provided the best protection against the formation of landslides (Phillips & Marden 1999). This was true both during normal periods of rainfall, and during the extreme rainfall event. Regenerating scrub and exotic pines 6–8 years old provided an intermediate level of protection, The greatest amount of damage occurred on pasture, and in exotic forest less than 6 years old (Marden & Rowan 1993). Similar work in hill country near Whatawhata (Waikato region), examined rolling (17–20°) to steep (over 30°) slopes, and found that a pasture catchment exported three times as much sediment as an adjacent native forest catchment. Beyond the immediate sediment outputs, landslide scars and tails can also generate ongoing sediment erosion for a further 1–2 years after heavy storms, after which they become stabilised and re-vegetated (Hicks et al. 2000). Streams draining native forest have lower suspended sediment loads, water temperatures, nutrient concentrations, and higher water clarities, than those draining pine forest and pasture (Quinn & Stroud 2002) (see Section 3.3).

Soil strength is improved by the presence of root structures that bind the soil together, and whose elasticity allows them to withstand large shear displacements before failure (Jones 2008). They also remove water from the soil, so that dense vegetation can act as a physical buffer against overland flow. Vegetation type also plays a role, with stands of the shrubby hardwood kanuka (*Kunzea ericoides*) providing a higher level of slope stability than pine stands for their first 9 years after establishment, and providing a similar level of protection after 16 years (Phillips & Marden 1999). Sediment is generated both during and following forest harvesting (Jones 2008) (Figure 6) (see also Whangapoua Estuary Section 6.1).

Figure 6: Clear-cut forest plantation. (Source; Dominic McCarthy, ARC.)

3.2.2 Arrival of suspended sediment at the coast

Several predictive models for estimating sediment yields from catchments have been developed to help in planning and management. Environment Waikato has created the model SedRate, which uses the relationship between measured river flows and suspended sediment concentrations at peak flows, to calculate the long-term average sediment load carried by rivers (Jones 2008). It does not include bed-load transport (which contributes about 3–10% of total sediment). Another model now available is the Hick's & Shankar model, which estimates suspended sediment yield based on mean annual rainfall and an 'erosion terrain' classification, which is based on data on slope, rock-type, soil and erosion processes, along with expert knowledge (Hicks & Shankar 2003, Hicks et al. 2004). This model can be used for any catchment throughout New Zealand. It should be noted that these are simply physical measures of sediment yield; they do not directly translate into the degree of ecological impact on coastal ecosystems.

The fate of suspended sediments on arriving in the marine environment depends on the coastal geomorphology. In many regions of New Zealand, estuaries and tidal lagoons act as giant sediment traps, and are especially vulnerable to sedimentation and its associated environmental and ecological effects. On more exposed coasts, where rivers discharge directly to the open sea, the suspended sediment is transported directly out onto the shelf, and may be dispersed over large spatial scales.

3.2.3 Estuaries

In estuarine environments, sedimentation effects over longer time scales are often captured in stratified sediment layers, and can be used to calculate sediment accumulation rates (SAR). Core sampling from numerous estuaries around New Zealand all show the same trend towards significantly increased sedimentation rates following large-scale deforestation (Table 1). Coromandel estuary examples include Wharekawa Estuary, with pre-Polynesian SAR of 0.09–0.12 mm yr^{-1}, rising to 3.0–7.2 mm yr^{-1} during

Table 1: Sediment accumulation rates (SAR) from various coastal regions in New Zealand (see Fig. 1 for locations). Numbers in brackets after the SARs are the years during which the estimates apply.

Region	Land-use practice	Pre-Polynesian SAR (mm/yr)	Post-Polynesian SAR (mm/yr)	Post-European SAR (mm/yr)	Reference
Wharekawa Estuary	Deforestation	0.09–0.12	3.0–7.2 (1880–1945)	5.0–8.0 (1945–1999)	Swales & Hume (1995)
Whangamata Estuary	Deforestation	0.01 (700 BP)	11 (after 1880)	5 (since 1940)	Swales & Hume (1984)
Whangapoua Estuary	Deforestation	0.03–0.08	0.12–0.13	0.89–1.5	Jones (2008)
Waitetuna Arm, Raglan Estuary	Deforestation	0.35	1.1 (since 1890)	2.5–8 (since 1990)	Swales et al. (2005)
Tamaki Estuary	Deforestation	0.11–1.6 (10 000 BP)	2.4	6.25 (1840 onwards)	Abrahim 2005
Papukura Estuary	Deforestation	0.2–0.5	0.8–1.6 (mid 1880s)	32.6 (since 1960)	Swales et al. (2002)
Waitamata Harbour	Deforestation	<1.5	2.5 (700–1100 BP)	3 (1841–present)	Hume & McGlone (1986)
Pauatahanui Inlet	Deforestation	1		2.4–3.4 (after 1850s), 4.6 since mid 1980s	Swales et al. (2005)
Wainui, Totaranui, and Awaroa Inlets	Deforestation	0.5–1.7	1.62–2.7	2.3–3.3 (within last 30 yrs)	Goff & Chague-Goff (1999)
Waikawa Estuary	Deforestation		1.5 (1878–1967)	3.1–10.7 (1967–2007)	Robertson & Stevens (2007)
Waituna Lagoon	Farming	0.05–0.6 (7000 BP-1960)		2.8 (since 1960)	Cadmus & Schallenburg, in press

catchment deforestation (1880–1945), and 5.0–8.0 mm yr^{-1} more recently (1945–1999) (an exotic pine production forest was established during this time) (Swales & Hume 1995); Whangamata Estuary, with pre-Polynesian (about 700 B.P.) SAR rates of about 0.01 mm yr^{-1}, increasing to 11 mm yr^{-1} after 1880 (Sheffield et al. 1995) due to clearance of relatively steep catchment and commercial forestry development, and estimated to be around 5 mm since the 1940s (Swales & Hume 1984); Whangapoua Estuary, with pre-Polynesian SAR rates of 0.03–0.08 mm yr^{-1}, increasing to 0.12–0.13 mm yr^{-1} following Maori occupation, and to 0.89–1.5 mm yr^{-1} following European forest clearances.

On the west coast, harbour-wide, Raglan Harbour SAR has averaged 0.3–0.5 mm yr^{-1} over the last 8000–6500 years (Swales et al. 2005). Following large scale deforestation of the catchment (1890–1920s), subsequent conversion to pasture, and more recent (1985–present) plantation forestry, sedimentation histories in two different arms of the harbour followed very different trajectories. In the Waitetuna Arm, pre-human SAR of 0.35 mm increased three-fold after deforestation, and has averaged 1.1 mm yr^{-1} since 1890. Pine pollen presence suggested that the SAR rate has further increased to 2.5 mm yr^{-1} since the early 1990s, with a maximum of 8 mm yr^{-1} at a site in Okete Bay (Swales et al. 2005). Conversely, in the larger Waingaro Arm of the harbour, indications are that long-term sedimentation has not occurred in at least the last 150 years (probably much longer). The cores also did not contain bracken pollens in association with native forest pollens (taken as an indicator of disturbance as a result of Maori slash and burn agriculture), nor the isotopes ^{137}C and ^{210}Pb (generated from Pacific Ocean nuclear tests in the 1950s). This was interpreted as evidence for sediment re-suspension by waves driven by the prevailing southwest wind (Swales et al. 2005), showing that sedimentation rates are dependent on the physical receiving environment, as well as the arrival of suspended sediment loads.

Around the city of Auckland, work in the Tamaki Estuary found early to late Holocene (the last 10 000 years) SAR rates to be about 0.11–1.6 mm yr^{-1}, when the surrounding catchments were vegetated in podocarp hardwood forests. Following Maori settlement and associated forest clearance, SAR rates increased to 2.4 mm yr^{-1}, and following European land clearances from about 1840 onwards, SAR increased to 6.25 mm yr^{-1}, with significant increases of heavy metals (Cd, Cu, Pb, and Zn) in the most recent layers (Abrahim 2005). In the Papukura Estuary, pre-human SAR rates ranged from 0.2–0.5 mm yr^{-1}; these rates increased three-fold to 0.8–1.6 mm yr^{-1} following European forest clearance and subsequent agriculture in the mid 1800s, and at the top of the estuary have averaged 32.6 mm yr^{-1} since 1960 (Swales et al. 2002). In the Mahurangi Harbour, following catchment deforestation (1850–1900), 3 metres of sediment has accumulated at the head of the harbour, 70% of this since 1900 (Swales et al. 1997). Infrequent floods were found to drive much of the erosion, with one-third of the total catchment erosion being generated from nine floods from 1953 to 1995. In Lucas Creek, in the upper Waitemata Harbour, rates increased from less than 1.5 mm yr^{-1} before human arrival, to 2.5 mm yr^{-1} during Polynesian forest clearance (700–110 BP), and then to 3 mm yr^{-1} after Europeans arrived, with associated logging, gum digging and land clearance (AD 1841 to the present (Hume & McGlone 1986)).

At the bottom of the North Island, 15 km north of Wellington, Pauatahanui Inlet sediment cores returned SAR estimates of about 1 mm yr^{-1} over the past several thousand years, increasing to about 2.4 mm yr^{-1} over the last 150 yr., with a further increase to about 4.6 mm yr^{-1} since the mid 1980s in the Horokiri subcatchment. The subcatchments size and stepness, combined with large-scale planting of pine forest since the 1970s, has resulted in an SAR of 10 mm yr^{-1} over the last two decades, twice the rate of elsewhere in the Pauatahanui Inlet. Harvesting of the 800 ha of pine forest over the next 10–15 years is likely to further increase this sedimentation rate (Swales et al. 2005). At the top of the South Island, coring in the Wainui, Totaranui, and Awaroa Inlets, inside the Abel Tasman National Park, Goff & Chague-Goff (1999) quantified a 1700 yr sediment record at sites that are now mature salt marsh, but were originally open tidal flats. Before European settlement, SAR rates ranged from 0.5 to 1.7 mm yr^{-1}, which increased following their arrival to 1.62–2.7 mm yr^{-1}, increasing in the last 30 years to 2.3–3.3 mm yr^{-1}. Associated with the European period were increased concentrations of Zn, and higher proportions of fine sediments. Also observed in the record were two “catastrophic saltwater inundation events” – namely tsunami – clearly recorded at about 1440 AD and about 1220 AD, across more than one site, with two less clearly defined ones at about 1855 BP and about 1600 BP (Goff & Chague-Goff 1999).

Further south again, in Waikawa Estuary (Figure 7), about 145 km sorth of Dunedin, the average SAR rate from 1878 to 1967 was 1.5 mm yr^{-1}, increasing to 3.1 mm yr^{-1} from 1967–1996, and to 10.7 mm yr^{-1} from 1996 to 2007 (Robertson & Stevens 2007). Half of the estuary's surface is now covered by soft mud. However, even before 1878 the upper estuary was covered with at least 0.5 metres of 'smooth grey mud', with few shell fragments. This was suggested to point to a period of very rapid sedimentation, perhaps resulting from land clearance in the mid 1800s (Robertson & Stevens 2007).

Figure 7: Waikawa estuary, Southland, and associated land use (pastoral farming and forest). The darker patches halfway up the channel are seagrass meadows, which extend subtidally to about 1.5 metres (sighted 2006). Associated fishes include juvenile leatherjackets. Sand flounders occur in high densities further up the harbour on the mud flats. (Source: LINZ website.)

In Waituna Lagoon, an intermittently open to the sea lagoon (13.5 km^2) just north of Invercargill, catchment run-off has been identified as one of the major stressors. Historically, a huge peat bog of about 200 km^2 stretching from the Fortrose Estuary to the New River Estuary surrounded the lagoon, giving it a characteristic clear brown humic stain, low nutrient levels, and low pH (Stevens & Robertson 2007). Only about 22 km^2 (11%) of the bog now remains, with the catchment dominated by intensive sheep, beef, and dairy farming. Sedimentation rates in the lagoon have risen from 0.05–0.6 mm yr^{-1} (7000 BP to 1960) to 2.8 mm yr^{-1} since 1960 (Cadmus & Schallenburg in press).

Direct run-off to the sea

In some areas, slope run-off carrying sediment may flow directly into adjacent coast zones (Figures 8, 9). For instance, the loess soils of Banks Peninsula are highly erodible, with a combination of deforested hills and periods of substantial run-off at the ends of long coastal embayments (Fenwick et al. 2003). A combination of this and large braided alluvial rivers with high sediment loads result in a continual supply of fine sediments to Pegasus Bay (Fenwick et al. 2003). Sediment is transported from south to north along the peninsula, especially when north-flowing coastal current, flood tidal streams, and southeasterly swells coincide (Dingwall 1974). Satellite imagery analysed by Cochrane & Male (1997) shows this to be a general pattern for the east coast of the South Island, while elsewhere suspended sediment movement is primarily offshore in a fanlike dispersal pattern with increasing dilution with seawater.

Figure 8: Aerial photograph of Mahia Peninsula, from the west, showing suspended sediment fringe around the land, taken 15 April 2005. (Source: Anna Madarasz-Smith, Hawkes Bay Regional Council.)

The Marlborough Sounds also provide direct inputs into the coastal system, with concerns expressed at the possible influence of exotic forestry on their magnitude. Fahey & Coker (1992) quantified sediment production from forest roads into Queen Charlotte Sound. Background rates of erosion were estimated at 300–600 t km^{-2} yr^{-1}. With 39 kilometres of road and 21 kilometres of forest track and firebreaks, about 2000 t of material was estimated to be removed by surface erosion each year (equivalent to 62 t km^{-2} yr^{-1}), which could increase to 7000 t (218 t km^{-2} yr^{-1}) at harvesting, with log landings adding a further 20%. Up to 200 t may have entered local marine embayments each year (Opua Bay), with the potential to raise suspended sediment concentrations to 1000 mg l^{-1}. Background concentrations were thought to be about 15–20 mg l^{-1}, rising to 1000 mg l^{-1} during storms (O'Loughlin 1980). In 1992, the

Marlborough Sounds contained 20 000 ha of pine, 6000 of these on slopes steeper than 25°, with soils with high clay content and low aggregate stability. In 1983, two large storm events a few months apart triggered numerous landslides on recently logged slopes, causing extensive damage (Fahey & Coker 1992).

Figure 9: Near-shore suspended sediment fringe due to erosion of near-shore seabed and step (formed in the Pleistocene, 5.1–1.81 million years ago) outwash gravels along the South Canterbury coast. (Source: Environment Canterbury.)

3.2.4 River plumes

In many regions of the country river mouths discharge directly to the open coast, where the freshwater flow creates a buoyant, low salinity river plume (Figures 10, 11). Typically, these contain large quantities of nutrients and sediments, and various pathogen levels (Robertson & Stevens 2006), depending in part on the number of livestock present in the catchment. In some regions significant amounts of terrestrial material, including leaves, twigs, branches, and entire trees may be carried into the near-shore environment, especially during flood events. Relatively little is known about the behaviour of river plumes in the New Zealand context, which vary widely in size, and may discharge into steep reflective gravel beach areas, semi-enclosed embayments, shallow shelf waters, and deep water (Robertson & Stevens 2006). They can cover large areas of coastal water. Large South Island east coast rivers have plumes that can extend northwards as bands of low salinity and discoloured water for at least 100 kilometres during high flow events, and merge with plumes from other rivers e.g. the

Clutha River merges with the Taieri and Kaikorai rivers (Gibbs & Adam 1982, Murdoch et al. 1990). Almost all of the fine sediment from these rivers is deposited tens to hundreds of kilometres northwards, e.g., fine sediment from the Clutha deposits in the lee or up-drift side of the Otago Peninsula (Carter 1986), while fine sediment from the Waitaki and Rakaia rivers deposits 100–200 km north on the up-drift side of Banks Peninsula (Gibb & Adams 1982).

Figure 10: Waimakariri river mouth, Pegasus Bay. Banks Peninsula can be seen in the background. Source: (Murray Hicks, NIWA.)

Further north, Hume & Nelson (1986) commented that LANDSAT (satellite) images showed fine sediment plumes from the Raglan, Aotea, and Kawhia harbours (west coast North Island) extending to 20 kilometres offshore following storm events. The clay fractions (less than two microns) of the inner shelf sediments directly offshore and north of these harbours are similar to the sediments inside the harbours, indicating that the source of these sediments are the readily erodible Oligocene (34–24 million years old) mudstones in the associated catchments.

Figure 11: River plume from Hurunui River mouth, North Canterbury. Source: (Bill Ballantine, Leigh.)

The Motueka River, which discharges into Tasman Bay, Nelson, has been the focus of a multi-year programme on the river, its catchment, and its influences via its plume into Tasman Bay (http://icm.landcareresearch.co.nz/). The surface salinity plume can extend more than 20 km into Tasman Bay, and after a major flood event with the appropriate wind directions can push north around Separation Point and into Golden Bay (Tuckey et al. 2006). An area of about 50 km^2 around the harbour mouth is contaminated by heavy metals (nickel and chromium) settling out from the plume, which has been traced back to a natural upper catchment mineral belt (Forrest et al. 2007). Concentrations strongly exceed sediment quality thresholds for probable ecological effects. Quantification of a number of indicators of terrestrial influence on seafloor sediments (organic carbon–nitrogen ratios, lipid biomarkers, trace metals, stable carbon and nitrogen isotope signatures in bivalves (shellfish) found an influence extending out at least 6 kilometres from the mouth, with the strongest influences in the first 2 kilometres (Forrest et al. 2007). Sites within 2 kilometres of the mouth were also dominated by mud, and cores also contained woody debris, leaf litter, and salt-marsh vegetation; while further out to sea coarser sediments dominated. They concluded that the overall plume effect was relatively localised, but also noted that during flood flows the river plume could extend tens of kilometres offshore.

3.2.5 An extreme example – Cyclone Bola and the east coast

In 1988 a 100 year storm, Cyclone Bola, hit New Zealand. Up to 900 m of rain fell in 72 hours (Singleton et al. 1989a, Sinclair 1993), resulting in rivers discharging several times their mean annual loads (Foster & Carter 1997). The heaviest rainfall was over steep hill country composed of highly erodible, soft Tertiary (65–1.8 million years old) siltstones and mudstones (Singleton et al. 1989b). Severe erosion caused river systems to aggrade rapidly, resulting in flooding of surrounding areas (Singleton et al. 1989). Analysis of satellite imagery showed an estimated 10–20% of the hill country in the east coast–Gisborne region to have experienced severe land-sliding in response to Cyclone Bola (Trotter 1988). The Waipaoa River, one of the main rivers in the region, ranks fourth in New Zealand for sediment yields (12.9 million tonnes yr^{-1}) (Griffiths & Glasby 1985), 97% of which is mud and fine sands (Adams 1980, Miller 1981, Griffiths & Glasby 1985). Associated with this, the annual sediment yield per km^2 of catchment is 5 836 tonnes, the fifth highest in New Zealand. During the 6 days of Cyclone Bola, 40 million tonnes passed into the marine environment. As described by Foster & Carter (1997), "*the continental shelf off Poverty Bay was inundated with mud*".

The suspended sediment concentrations were such that it was thought to form a subsurface plume (i.e., on the seafloor rather than the surface) that moved and dispersed under the influence of gravity and shelf currents (Foster & Carter 1997). Subsequent observations by fishers and divers suggested that this layer, up to 2 metres thick, as measured against a shipwreck and lobster pot lines near reefs throughout the bay, extended right across the inner to middle Poverty Bay shelf. This layer was mobile, with observations of reef areas being covered and uncovered. This layer smothered the resident benthic assemblages, and left the area strongly depleted in species numbers and diversity (Battershill 1993). While the effects on coastal fisheries were not documented, the effects were likely to have been profound, both directly through mortality and/or dispersal of species, and by the degradation of important seafloor habitat and associated prey assemblages. On a longer time scale, the modern rates of sedimentation to the seafloor in this region are now almost five times higher than before European deforestation in the late 19th century (see Figure 12 for an example of current day sediment plumes).

Figure 12: Satellite (SEAWIFS) image of the Bay of Plenty and east coast North Island, showing large coastal areas with suspended sediment evident. White Island ash plume in upper centre of image. (Source: Lionel Carter, NIWA.)

3.3 Eutrophication via elevated nutrient loads – generation and delivery to the coast

3.3.1 The start of the chain – freshwater systems

One of the consequences of changing land use, and significant increases in the numbers of animals living on the land (e.g., sheep, cattle, and humans) is a significant increase in the concentrations of nutrients entering waterways (in particular, nitrogen and phosphorus), much of which eventually reaches estuarine and coastal ecosystems. New Zealand's farming economy has resulted in strong effects on freshwater systems (lakes, rivers, and streams) with significant impacts on water quality and the associated fauna and flora inhabiting them. Lowland rivers in agriculturally developed areas have been subjected to high nutrients, turbidity, and faecal contamination, leaving them in a poor condition (Parkyn et al. 2002). Streams in areas of dairy farming, especially where poor practices of shed effluent disposal have been used, are in particularly poor condition, and the intensification of farming associated with dairying in general has also been related to increasing levels of nutrients, sediments, and faecal bacteria (Parkyn et al. 2002). At the national level, streams sitting in or near native forest generally have good water quality, with many examples of streams originating in forested headwaters having healthy invertebrate communities, while the same streams further down the catchment passing through increased pastoral development have invertebrate communities low in diversity and dominated by high pollution tolerant species. In pasture-dominated catchments (over 50% cover), most lakes have clarity levels and nitrogen and phosphorus concentrations that breach water quality guidelines. Pasture streams also have elevated levels of suspended sediment and turbidity, and lower water clarities (Dons 1987, Smith et al. 1993, Quinn et al. 1997, Quinn & Stroud 2002) than native streams, due to increased run-off, erosion, and bank instability. They also have higher levels of nitrogen and phosphorus nutrients (Cooke 1979, Wilcock 1986, Cooper et al. 1987, Cooper & Thomsen 1988, Quinn et al. 1997, Quinn & Stroud 2002), resulting from increased run-off, eroded sediment, and subsurface leaching losses carrying excess nutrients from fertilisers, nitrogen fixation, and stock excreta on pastures, as well as inputs from fertiliser drift, and stock excreta in waterways (Quinn & Stroud 2002); and higher faecal coliforms (Smith et al. 1993), *E. coli* and pathogens (Donnison & Ross 1999) than native streams; and in many cases exceed water quality guidelines. Vant (1999) found nitrogen yield in eight large Waikato catchments strongly correlated with dairy cow stocking density.

The impact on freshwater aquatic assemblages has been profound. As land is cleared and grazed, there is a reduction in shade leading to algal blooms and increased temperature (Quinn et al. 1997, Rutherford et al. 1997, 1999), a reduction in organic matter inputs (e.g., leaves and twigs) that are habitat and food sources (Scarsbrook et al. 2001), increased nutrients adding to in-stream plant growth, increased sediment inputs (Quinn & Stroud 2002), changes in stream morphology and wood inputs (Davies-Colley 1997), deepening and straightening of channels that increase stream gradients, reduce stream length and habitat diversity (Williamson et al. 1992), and increased flow yield, variability and surface runoff (Dons 1987, Fahey & Rowe 1992). These impacted waterways ultimately empty into the coastal marine environment.

3.3.2 Nutrient enrichment in the marine environment

Our understanding of the impacts of eutrophication in New Zealand estuaries, embayments, and the near-shore environment (we would suggest) is much less advanced than for freshwater systems. What work there is, appears to have largely been focussed on showing that N and P are often elevated due to adjacent land-based activities. The best example of monitoring is the long-term data series generated by the Auckland Regional Council, which consists of a comprehensive water quality monitoring network across 27 estuary and near-shore sites, from 1987 to the current day (Scarsbrook 2008). Inner harbour sites tend to have the poorest water quality, while outer harbour and coastal sites are rated as relatively good (Scarsbrook 2008). Across the overall region, there are significant improving trends in levels of faecal indicator bacteria, total suspended sediments, total phosphorus, soluble reactive phosphorus, and nitrate, mostly consistent with decreased anthropogenic pressures. Strong temporal correlations were observed between streams and estuaries in the region, suggesting that the water quality of streams is a major determinant of water quality at adjacent inner harbour sites. The role of climate was also found to be important, with strong temporal links between the Southern Oscillation Index (SOI) and patterns of water quality, particularly for temperature and nitrogen concentrations. Over the time series, three sites in the Manukau Harbour have stood out from all others, with respect to nitrogen and phosphorus, being heavily affected by discharges of treated sewage water from the Mangere Wastewater Treatment Plant. Following the decommissioning of the Mangere oxidation ponds in 2002, dramatic improvements have occurred, especially in levels of ammonical nitrogen, total phosphorus, and suspended sediments.

We could find little else published around the temporal monitoring of N, P, and other nutrients in estuarine and coastal environments.

3.4 Pollution – heavy metals and other chemicals

Monitoring by regional councils and others has shown that heavy metal concentrations are often strongly locally elevated around built up human settlements. Most of these heavy metal ‘hotspots’ occur in upper estuarine areas, and are rather localised in their spatial extent – a finding supported by overseas research. Kelly (2007) reviewed heavy metal monitoring (copper, lead, zinc, and "where required, polycyclic aromatic hydrocarbon (PAH)"), across 72 sites in the Auckland region, starting in 2002, and commissioned by the Auckland Regional Council (ARC). Sites were measured at 2 to 5 year intervals, depending on metal concentrations. The highest concentrations of copper, lead, and zinc were obtained from estuarine sites adjoining the older urban catchments of Waitakere, Auckland, and Manukau cities i.e., Henderson Creek to Cox's Bay along the southern shores of the Waitemata Harbour; the upper reaches and side-branches of Tamaki Estuary; and Mangere Inlet. Sites with the highest heavy metal concentrations were found to have the highest rates of increase in heavy metal concentration, especially for copper and zinc, while lead concentrations were found to be more variable over time, reflecting its removal as a petrol additive. Overall, with the exception of Mangere Inlet, levels of all three metals were found to be below threshold effect levels (i.e., TEL sediment quality guideline values) in the Manukau Harbour, and Orewa and Weiti estuaries. Copper and zinc concentrations were stable or slowly increasing. For example, Pahurehure Inlet zinc concentrations increased by 27% between 1998 and 2005, while concentrations in the Weiti estuary increased by 30% over the same period. Copper concentrations were found to be slightly above the TEL thresholds at a number of upper Waitemata sites, while lead and zinc concentrations were below TEL thresholds except at Hellyers Creek. Zinc concentrations are increasing rapidly in Lucas Creek, and are likely to soon exceed the TEL threshold. Kelly (2007) stated that a strong relationship was apparent between copper, lead, and zinc concentrations and benthic community structure, indicating that the current levels of contamination (or a covariate of copper, lead, and zinc) are affecting the ecological function of urban estuaries. In general, the spatial pattern of ecological condition reflected levels of contamination.

Similar work by the Wellington Regional Council, across 17 sites in Wellington Harbour, has found elevated concentrations of lead, mercury, and to a lesser extent copper and zinc, with concentrations exceeding sediment quality guidelines in some areas, such as adjacent to the commercial port wharves (Stephenson et al. 2008).

The scientific literature around heavy metals and chemical pollutants, and their impacts at the cellular and individual organism level, is extensive and detailed. The scope of this review does not allow us to examine these areas in detail. However, for an exhaustive and intensive review, the reader is directed to Grant & Hay (2003), who looked at these issues in terms of intertidal shell-fish depletion in the greater Hauraki Gulf Marine Park. They concluded that "*While acknowledging that there are significant knowledge gaps, in general it appears that the occurrence of potential stressors at high levels are relatively rare and localised*" and that "*there is a general trend of increased risk of anthropogenic contaminants in inter-tidal zones in enclosed estuarine areas as opposed to open coastal environments*".

3.5 Freshwater extraction

An emerging issue, both in New Zealand and internationally, is the increasing level of extraction of freshwater before it reaches the marine environment. Such allocations now require resource consent in New Zealand, and so estimates of total removals versus overall estimated flows should be possible at both regional and national levels. Effects on coastal fisheries are likely to be expressed through changes in river plume extents.

4. MECHANISMS OF IMPACT

In this section we discuss the general mechanisms of impact, based on published information in the scientific literature. As many of these mechanisms are synergistic, there is some cross-over between different sections, and we emphasise that in reality these processes may operate simultaneously on fish populations and fisheries. Here we focus on general published descriptions, including New Zealand systems and species where available. Given a lack of marine examples, we have also included freshwater examples as a probable proxy for effects in the marine environment. Specific New Zealand fisheries species examples are covered in later sections in their own right.

4.1 Sedimentation and suspended sediments

Increases in sedimentation to the coastal zone can produce a wide range of effects, both from deposition of fines on the seafloor, and as suspended sediments in the water column. Ongoing re-suspension and deposition events (e.g., by storms and fishing gears) may shift sediments between these two states. Direct effects on species include the clogging of gills and reductions in visual foraging efficiencies, with associated acute and/or chronic impacts, e.g., immediate physiological stress and reduced growth rates and reproductive fitness. Indirect effects include the loss of important nursery habitats such as biogenic habitat formers, and reductions in prey assemblage abundances. These effects do not act in isolation from each other, and may produce additive or multiplicative outcomes.

4.1.1 Direct physical and physiological effects

Invertebrates

Elevated levels of suspended sediments can reduce the diversity and abundance of pelagic and benthic invertebrates for both freshwater and estuarine systems (Quinn et al. 1992, Harding et al. 2000, see reviews: Thrush et al. 2004, Gibbs & Hewitt 2004, Lloyd 1987, Newcombe & MacDonald 1991, Bash et al. 2001, Berry et al. 2003) by abrading, clogging and smothering organisms; reducing interstitial spaces; and reducing food supply and quality through decreased light attenuation and hence aquatic algae and plant productivity. Other impacts include reduction in feeding rates, changes in behaviour, and increased susceptibility to diseases (Newcombe & MacDonald 1991). Suspended sediments have been found to be acutely toxic to young-of-the-year amphipods (Forbes et al. 1981), while Schwarz et al. (2006a) found decreased survival rates for the gammarid amphipod *Aora* sp. at high suspended sediment concentrations. This may have implications for juvenile fish health, as recent work in northern New Zealand estuaries shows amphipods are an important component of juvenile fish diets (M. Lowe, Leigh Marine Laboratory and NIWA, unpubl. data).

Suspension feeding bivalves are especially vulnerable through their water filtering activities. Work on their physiological responses to increasing suspended sediment concentrations have shown decreases in clearance rates (Bricelj & Malouf 1984, Ward & MacDonald 1996, Bacon et al. 1998), oxygen consumption (Grant & Thorpe 1991), and growth (Bricelj et al. 1984, MacDonald et al. 1998). Bivalves may respond to high suspended sediment loads by reducing their pumping rates (Foster-Smith 1976) and rejecting excess filter material as pseudofaeces (Turner & Miller 1991, Hawkins et al. 1996), resulting in decreasing energy returns as suspended sediment loads increase. However, the responses of filtering bivalves to low sediment concentrations vary. Blue mussels (*Mytilus edulis*), surf clams (*Spisula subtrucata*), and eastern oysters (*Crassostrea virginica*) showed increasing growth rates when silt was added to high algal concentrations, presumably through some mechanism that enhanced their filtering efficiencies (Kiorboe et al. 1981, Urban & Langdon 1984). Conversely, northern quahogs (*Mercenaria mercenaria*) decreased their algal ingestion with increasing sediment loads (Bricelj & Malouf 1984) with no detectable change in growth rates relative to quahogs fed only algae (Bricelj et al. 1984). European oyster (*Ostrea edulis*) summer growth rates increased at low concentrations of sediment re-suspension, but were inhibited with increased sediment deposition (Grant et al. 1990). Adult northern quahogs and eastern oysters exposed to sediments with high silt-clay content suffered reduced growth and lower survival, respectively (Pratt & Campbell 1956, Kirby 1994).

Shellfish do have some natural resilience to such impacts. The green-lipped mussel (*Perna viridis*) was shown to be able to survive in suspended sediment concentrations of up to 1 200 mg l^{-1} without mortality over a period of 4 days (96 hours) (Shin et al. 2002), an adaption attributed to a high rejection efficiency of mucus-bound strings of particulate matter (except for the finest particles) by its labial palps in the mantle cavity (Seed & Richardson 1999). However, later work by Cheung & Shin (2005), using suspended sediment of less than 500 microns (half a millimetre) in size, at concentrations ranging from 0 to 1000 mg l^{-1}, over a experimental period of 14 days, found significant damage to occur to the feeding cilia. This varied with suspended sediment concentrations and exposure time, with no sign of recovery after 28 days of mussels being held post experiment in clear, filtered seawater. A second experiment using suspended sediments of less than 63 microns, 125–250, and 250–500 micron size fractions, at 600 mg l^{-1} over 14 days, found the damage of cilia was greater in the larger sediment size classes.

In the New Zealand context, studies by Lohrer et al. (2004), Norkko et al. (2002), & Thrush et al. (2004) found that deposition of suspended sediments can alter substrate composition, leading to increased clay and silt content, affecting the distribution of infaunal and epibenthic species. Addition of as little as 3 mm of terrestrial sediment experimentally deposited onto the seafloor surface was reported to alter soft sediment macrobenthic community structure, while deposition of 7 mm reduced individuals and species abundances by 50% (Lohrer et al. 2004, 2006). Larger bivalves were less affected than small ones, as were deeper dwelling ones. Gibbs & Hewitt (2004) reviewed these and other experiments designed to assess the impacts of sedimentation on benthic assemblages, and produced the following guidelines for likely impacts.

- The thicker the layer of mud, the more animals will be killed and the longer recovery will take. This will affect both the number of species and the number of animals within each species – some species are more sensitive than others.

- If mud is washed down a stream to a tributary estuary or embayment results in a mud layer greater than 2 cm for more than 5 days, all resident animals except mobile crabs and shrimps will be killed due to lack of oxygen.

- Mud thickness of around 5 mm for more than 10 days will reduce the number of animals and number of species, changing assemblage structure.

- Frequent deposition of mud, less than 5 mm, may still have long-term impacts that can change animal communities.

The heart urchin (*Echinocardium australe)*, a large burrowing deposit feeder, was adversely affected after 3 days in suspended sediment concentrations of more than 80 mg l^{-1}. Burial times and death rates increased with increasing exposure to suspended sediments (Gibbs & Hewitt 2004). The deposit feeding polychaete *Boccardia syrtis* was similarly adversely affected at concentrations above 80 mg l^{-1} after 9 days. Feeding rates decreased over time, with the greatest decreases in the highest concentration treatments. Wedge shells (*Macoma liliana*) were adversely affected at concentrations above 300 mg l^{-1}, after 9 days exposure, and by 15 days of exposure at the highest concentration levels most had died or were lying exposed on the sediment (Gibbs & Hewitt 2004).

Finfish

Most of our current knowledge on the effects of suspended sediments on fish are based on freshwater (in particular salmonid) species (see reviews: Bruton (1985), Lloyd (1987), Newcombe & MacDonald (1991), Newcombe & Jensen (1996), Kerr (1995), Moore (1977), Bash et al. (2001), and Wilber & Clarke (2001). Most existing information of the effects of suspended sediment is based on acute exposure laboratory experiments, with little empirical information available on chronic responses to high concentrations for extended periods, especially for marine species (Au et al. 2004), or under natural field conditions. Research on behavioural responses of estuarine fish to suspended sediment plumes is also largely missing. Here we use freshwater fish examples as a surrogate for marine species, as well as those marine fish examples that exist. Studies have used a mixture of nephlometric turbidity units (NTU) and actual suspended sediment concentrations; the two are not directly comparable through conversion to a common metric. Therefore, we have reported these studies using the same units as the authors.

Suspended sediment response categories for finfish comprise the following – none, behavioural, sub-lethal, and lethal. The behavioural category includes alarm reaction, abandonment of cover, avoidance response, and impaired homing (Newcombe & Jensen 1996). Sub-lethal effects include reductions in feeding rates/success, reduced growth rates, delayed hatching and reduced fish density, and habitat degradation; along with physiological responses such as changes in blood physiology, gill structure, increased respiration rates, and coughing (Wilber & Clark 2001).

Behavioural responses

Short term pulses of sediments (about 1 hour) have been shown to disrupt feeding behaviour of salmonids at turbidity levels as low as 20 NTU (Berg 1982), through reducing the reactive distance for visual feeding fish species (Vinyard & O'Brien 1976, Gardner 1981, Berg & Northcote 1985, Barrett et al. 1992, Confer et al. 1978), or by reducing fish feeding rates (Berg & Northcote 1985, Redding et al. 1987, Sigler et al. 1984, Gregory 1993). The feeding of juvenile coho salmon (*Oncorhynchus kisutch*) dropped by 45% at a turbidity of 100 NTU (Reid 1998), while reduced growth rates associated with increased suspended sediments have been documented for Artic grayling (*Thymallus articus*) (McLeay et al. 1987) and coho salmon and rainbow trout (*Oncorhynchus mykiss*) (Sigler et al. 1984).

Newly emerged fry (just post-larval fish) appear to be more susceptible to even moderate turbidity levels of 25–50 NTU, with reduced feeding and increased emigration recorded (Sigler et al. 1984). However, suspended sediment may also enhance the visual contrast of prey items, increasing overall feeding rates as reported for larval Pacific herring (*Clupea pallasi*) (Boehlert & Morgan 1985). Increased turbidity has also been reported to enhance feeding motivation due to reduced risk of predation while foraging for Chinook salmon (*Oncorhyncus tshawytscha*) at turbidity levels of 35 to 100 NTU (Gregory & Northcote 1993). Turbidity has also been shown to reduce the avoidance responses of juvenile Chinook salmon to bird and fish predator models (Gregory 1993). Thus, refuge from predators may constitute a survival advantage which negates the negative effects of reduced feeding and growth rates for some species.

Although not all fish avoid turbid waters, elevated suspended sediments often induce avoidance reactions and may modify natural movements and migrations by removing visual cues by which fish maintain position (Berg & Northcote 1985, Sigler et al. 1984, Lloyd et al. 1987, Berg, 1982). McLeay et al. (1984) found that Arctic grayling were displaced throughout a 6 week experimental period at suspended sediment concentrations of 300 mg l^{-1} or greater, while Servizi & Martens (1992) estimated that the avoidance threshold for juvenile coho was 37 NTU in the vertical plane. Berg & Northcote (1985) reported that for juvenile coho, short term pulses of high turbidities (30–60 NTU) broke down dominance hierarchies and territories were not defended, with fish relocating downstream to undisturbed areas. It was suggested that frequent short term pulses could thus decrease growth and feeding rates, and may affect overall mortality (Berg 1982).

Increased foraging time by Pacific cod (*Gadus macrocephalus*) for mysids has been recorded with increasing turbidity levels (.099 – ~59 NTU; Meager et al. 2005). Feeding in adult Atlantic croaker (*Micropogonias undulates*) and pinfish (*Lagodon rhomboids*) was also reduced in turbid water (Minello et al. 1987), as was the feeding of silverside (*Atherina breviceps*) at high turbidity conditions (120 NTU). This was thought to be a result of decreasing reactive distance of the fish to their planktonic prey, which can occur at turbidities as low as 28 NTU (Hecht & van der Lingen 1992). Long term reduced feeding rates could have serious consequences for juvenile fish affecting overall condition, recruitment, survival and year class strength.

However, turbidity effects on fish may vary, depending on the fishes' search volume, foraging strategy, and the influence of turbidity on prey behaviour to detect and avoid predators (Meager et al. 2005, Macia et al. 2003). When larval striped bass (*Morone saxatilis*) were fed primarily copepods, feeding rates declined at suspended sediment concentrations of 200 and 500 mg l^{-1}, but no change in feeding rate was noted when the prey item was the slower moving cladoceran *Daphinia pulex* (Breitburg 1988). Research in South African estuaries found fishes collected from turbid to relatively clear water showed little difference in both abundance and condition factor. However, stomach content analysis revealed a change in feeding strategy from highly efficient visual feeding in clearer conditions, to a more non visual 'encounter rate feeding' mode for turbid conditions (Hecht & Van der lingen 1992, Marais 1984).

Sub-lethal effects

Fish can tolerate short episodes of extremely high levels of suspended sediment by intensified mucus production in the gills, a response to fine particles coating the fishes' respiratory epithelia, causing hyperplasia. This effectively reduces the capacity for oxygen transfer, leading to respiratory stress (Bergstedt & Bergersen 1997, Kerr 1995). At very high turbidities, sediment-clogged gills cease to function and fish die from a combination of anoxemia and carbon dioxide retention (Ritchie 1972). However, in most cases elevated suspended sediments have sub-lethal effects on freshwater fish. Lethal concentrations of suspended sediments have been reported for a number of northern hemisphere fish species and are generally over 500g/m^3 (Alabaster & Lloyd 1980, Newcombe & McDonald 1991).

Reduced respiratory efficiency can result in increased ventilation rates to compensate, as has been recorded for green sunfish (*Lepomis cyanelllus*) under highly turbid conditions (Horkel & Pearson 1976). Gill flaring (Berg 1982, Berg & Northcote 1985) and increased cough frequency has been shown for juvenile coho salmon (Newcombe & Macdonald 1991). Cough frequency for coho was elevated eightfold over control levels at 240 mg l^{-1} (30 NTU) (Servizi & Martens 1992). 'Surface gulping' has also been recorded for juvenile coho salmon showing signs of anoxia (Berg & Northcote 1985) while suspended sediment concentrations of over 100 mg l^{-1} caused Arctic grayling to surface (McLeay et al. 1987). Other effects include increased fin rot and body abrasion (Ritchie 1972), paler coloration (McLeay et al. 1984), and delayed maturation (Reynolds et al. 1988).

Longer term chronic exposure can lead to haematological compensation for lost respiratory efficiency. Changes in blood physiology such as elevated levels of blood sugars (Servizi & Martens 1992), plasma glucose (Servizi & Martens 1987), microhematocrit (packed red blood cell volume), haemoglobin concentrations, red cell counts (Redding et al. 1987, Appleby & Scarratt 1989) and cortisol levels (Schreck 1981) have been recorded with increasing suspended sediment concentrations. For adult sockeye salmon (*Oncorhydus nerka*), plasma glucose levels increased 39% and 150% as a result of exposures to 500 and 1,500 mg l^{-1} respectively (Servizi & Martens 1987). Decreased tolerance rates to disease and time to death as a result of other environmental stressors have also been recorded for elevated suspended sediment concentrations (Appleby & Scarratt 1989, McLeay et al. 1984, Redding et al. 1987).

Extensive bioassays of suspended sediments on estuarine fish have been conducted by Sherk et al. (1974, 1975), O'Connor et al. (1977), and Neumann et al. (1982) over periods of up to 14 days. Sub-lethal effects of fuller's earth suspensions were measured by blood cell counts, haemoglobin concentrations, blood ionic composition, carbohydrate utilization, and gill histology. Frequent sub-lethal responses included, increased red cell counts, haematocrit, and haemoglobin concentrations in the peripheral blood. Results were consistent with fish deprived of oxygen (O'Connor et al. 1977). For white perch (*Morone Americana*), exposure to 650 mg l^{-1} of fuller's earth for 5 days resulted in a 30% increase in microhaematocrit, haemoglobin concentrations and red blood cell counts, relative to control groups (O'Conner et al. 1977). Similar responses were observed in haematocrit levels of hog-chokers (*Trinectes maculates*) and striped killifish (*Fundulus majalis)* exposed for 5 days to Fuller's earth concentrations of 1 240 and 960 mg l^{-1} respectively. Increased haematocrit was also recorded for striped bass which were exposed for the longest duration time of 14 days at a concentration of 1,500 mg l^{-1} (Sherk et al. 1974, O'Connor et al. 1977). However, oyster toadfish (*Opsanus tau),* which inhabit the turbid sediment-water interface, showed no significant respiratory responses to fuller's earth or natural sediment suspensions (O'Connor et al. 1997).

At high suspended sediment concentrations, white perch experienced gill tissue disruption and intensified mucus production, leading to respiratory stress (O'Conner et al. 1977). Larger particles were trapped by gill lamellae and stopped the passage of water leading to asphyxiation (Sherk et al. 1974). A further longer term study (6 weeks) on juvenile green grouper (*Epinephelus coioides*) by Au et al.

(2004) showed damage to gill structure, including epithelium lifting, hyperplasia in the pillar system, and reduction of epithelial volume were strongly correlated to suspended sediment concentration which ranged from 0 to 2000 mg l^{-1}.

Lethal responses

Mortality curves have been generated for six estuarine species by Sherk et al. (1974, 1975) using fuller's earth, while other species were tested for suspended sediment tolerances. Tolerant species included mummichog (*Fundulus heteroclitus*), striped killifish, spot (*Leiostomus xanthurus*), oyster toadfish, hog-choker, and cusk eel (*Rissola marginata*), all found at the sediment water interface. White perch, bay anchovy (*Anchoa mitchilli*), juvenile Atlantic menhaden (*Brevoortia tyrannus*), striped bass, Atlantic croaker, and weakfish (*Cynoscion regalis*) were classified as sensitive, having no particular habitat preference. Juvenile bluefish (*Pomatomus saltatrix*), Atlantic silversides (*Menidia menidia*), and age-0 white perch were regarded as highly sensitive, with the latter two showing the most sensitive lethal responses to suspended sediments, exhibiting 10% mortality at concentrations less than 1 000 mg l^{-1}, for 1 and 2 days respectively.

In summary, the effects of suspended sediment concentrations on fish are dependent upon synergistic factors including duration of exposure, frequency, magnitude, temperature, and other environmental variables (Servizi & Martens 1992), with responses varying greatly between species and developmental stages. The effects of these responses can ultimately compromise fish health, reproduction, year class strength, and distribution of adult populations.

4.1.2 Indirect effects of sedimentation on important fisheries habitats

A number of plant and animal species exist in sufficient densities, and/or with sufficient structural complexity, that they provide important habitat to many other species, and can be referred to as 'habitat-formers'. Loss of these species, and the associated functions that they provide for other species, can have profound effects on the functioning of the wider ecosystem, e.g., through reduction in nursery habitats. Examples of such species from temperate regions include plants such as seagrasses, seaweeds/kelps, and maerl/rhodoliths; and animals such as bivalves, oysters, sponges, and bryozoans. These are vulnerable to sediment driven mechanisms, as discussed in the previous section. For plants, the most dominant effect is the reduction in light levels from reduced light penetration through the water column, combined with siltation of the light-receiving surfaces of the plant itself. Additional effects can include the siltation of surfaces that are required by settling spores to grow, and abrasion of plant surfaces during periods of high storm and wave energies. Impacts on grazers may also occur; for instance, grazing by the limpet *Patella vulgata* was reduced by 35% with the addition of a 1 mm thick layer of sediment (equivalent to 50 mg cm^{-1}), while at a load of 4 mm total inhibition occurred, along with associated mortalities (Airoldi & Hawkins 2007). Herbivorous organisms such as limpets are often scarce in areas with high sediment loading (Airoldi & Virgilio 1998, Pulfrich et al. 2003, Schiel et al. 2006).

As with other stressors, the relative role of sedimentation in habitat loss is often not well known at the ecosystem level. Seagrasses are a good example, and are known to be a very important habitat component of estuarine and coastal ecosystems. Green & Short (2003) documented 170 000 km^2 to have been surveyed in some form, and tentatively suggested a world-wide global extent of 500 000 km^2. Numerous studies and accounts from many countries and regions consistently identify a long-term, world-wide trend of seagrass decline, about 70% of which can be assigned directly to human-induced disturbance (Short & Wyllie-Echeverria 1996). Less information is available on degradation caused by indirect effects (Duarte 2002). During the 1990s alone, estimated global seagrass loss was 12 000 km^2 (Short & Wyllie-Echeverria 1996), representing about 7% of the known extent (Green & Short 2003). Longer term data series are rare. Lortze et al. (2006) estimated that 65% of seagrasses have been lost across 12 temperate ecosystems. Causes are varied. In the 1930s a wasting disease caused by the pathogenic slime mould *Labyrinthula zosterae* (e.g. Den Hartog 1987) resulted in a catastrophic die-back of eelgrass (*Zostera marina*) meadows along the North Atlantic coast, and an almost 90% loss of beds in the North Atlantic of western Europe (Airoldi & Beck 2007). Some beds progressively recovered, but substantial areas remain lost from most beds, with subtidal beds being most affected. For example, Danish beds in 1900 totalled some 6 726 km^2, of which 93% was gone by 1940. Since 1960 slow recovery has occurred, and bed extents are now at about 20–25% of 1900 levels (Airoldi & Beck 2007). The greatest loss was from deeper beds, with the vertical distribution being reduced by about 50% during the 20th century, from 11.1 to 5.6 m in sheltered areas, and 8 to 2.5 m in exposed areas (Hemminga & Duarte 2000, Baden et al. 2003). In the Wadden Sea, seagrass decline had two phases (Reise 1994), the first being an acute decline in the 1930s from wasting disease, from which most subtidal beds did not recover, followed by a more gradual decline beginning in the 1960s, mostly driven by eutrophication. Seagrass cover dropped from 150 to 1–2 km^2, along with the disappearance of many seagrass associated species (Wolff 2000).

Many anthropogenic factors are considered responsible for the ongoing degradation and decline of seagrasses in Europe as well as globally (reviews by Short & Wyllie-Echeverria, Davison & Hughes 1998, Hemminga & Duarte 2000, Duarte 2000, Green & Short 2003). The most important are poor water quality from pollution, eutrophication, and excess sedimentation (Airoldi & Beck 2007).

On rocky reefs, seaweeds/kelps are important habitat-formers (Schiel & Foster 1986), and also support diverse and productive assemblages of small mobile invertebrates that contribute about 80% of energy flow and materials through rocky reef animal communities (Taylor 1998). The lower limits of seaweed distributions are thought to often be set by light availability (Spalding et al. 2003), with 2% of surface irradiance being suggested to be the lower limit for kelp (Markager & Sand-Jensen 1992). Declines of water clarity through time have been matched by reductions in maximum depth limits of seaweeds on rocky reefs (Lumb 1989, Kautsky et al. 1996). Sedimentation directly onto the seabed may also affect seaweeds directly (Airoldi 2003). Effects may include preventing the attachment of kelp spores (Devinny & Volse 1978), with experiments showing that settled sediments may suppress seaweed recruitment and/or favour turfing forms (Chapman & Fletcher 2002, Gorgula & Connell 2004, Schiel et al. 2006).

While globally there is little evidence of widespread impacts in declines in water quality on global kelp forests, there is also little long-term data on water quality on rocky reefs (Steneck et al. 2002). There has also been a strong focus on surface-canopy-forming species such as *Macrocystis*, which may have diverted attention from species which are affected by reductions in light transmittance (R. Cole, NIWA, pers. comm.). Cole & Babcock (1996) described a protracted die-back of *Ecklonia radiata* following dense phytoplankton blooms in northeastern New Zealand in 1992–93, and dieback was also noted at Goat Island, Leigh, in 1982–83 following similar blooms (A. MacDiamid, C. Battershill, pers. comm., via R. Cole). The 1992–93 event displayed a progressive mortality pattern up the reef, consistent with light limitation as the driving mechanisms (Cole & Babcock 1986).

4.2 Eutrophication

Eutrophication, through increasing nutrient concentrations, produces a cascade of effects in marine ecosystems (Levin et al. 2001). Heavy nutrient loading stimulates the production of phytoplankton and algal production, leading in turn to an increase in organic inputs to the seabed, and reduces oxygen availability. Macroalgae respond positively (up to a point), seagrass species are adversely affected, and light levels can be reduced, reducing the euphotic zone (the depth range over which plants can photosynthesise before light levels become too low). Diverse benthic communities may disappear and be replaced by ones dominated by deposit-feeding annelids (Sardá et al. 1998). Tracking of eutrophication effects in Scotland showed a replacement of seagrass by green algae, along with a change from a crustacean dominated assemblage supporting wading birds, to a benthos-poor algal matt without birds (Raffaelli 1999). Similar processes were seen in the Baltic Sea in a number of places, leading to a loss of more than 40 macrophyte species, all replaced by a single species of brown filamentous alga (Zmudzinski 1997, Jannson & Dalberg 1999). Associated with this was a drop in associated in-faunal invertebrate species, important as prey for fish. Fish spawning (perch and pike) grounds were also lost with the disappearance of plants.

An overall effect of eutrophication is a reduction in trophic transfer between benthic in-fauna and bottom-feeding fish (Jannson & Dahhlberg 1999), as system diversity is reduced to a few tolerant species which may not provide the nutritional values needed by fish. For example, in the northern Baltic Sea a gastropod (*Hydrobia*) replaced a bivalve (*Macomona*) as the primary dietary item of a sand goby. Up to 90% of the gastropods passed alive through the gobies' gut, in contrast to complete digestion of *Macomona*, strongly suggesting that the food value of this gastropod was minimal (Aarnio & Bonsdorff 1997). Powers et al. (2005) documented a similar story for the Neuse River Estuary, in North Carolina. Depletion of bottom water oxygen from eutrophication led to about 90% decline of the clam *Macomona balthica*, a key prey item for fishes and crabs. Associated with this was a shift in croaker (*Micropagonias undulatus*) diet from clams to less nutritional prey items, such as plant and detrital material. Work in the same system by Eby et al. (2005) found that the physiological condition, individual growth rate, and population growth of croaker and spot in the same Neuse River system were lower in the year of severe hypoxia than in two years of less intense summertime oxygen depletion, and suggested that this showed that switching to alternative preys was not adequate to maintain high fish production. Related mass balance modelling (working out energy flows) found that benthic in-fauna production was sufficient for the energy demands of demersal fishes and blue crabs before a large scale hypoxic event in 1997, but not after the event.

4.2.1 The filter of eutrophication

Cloern (2001) reviewed the concept and consequences of coastal eutrophication, and suggested that a 'filter' of different influences existed that determined how ecosystems responded. He noted that the disturbance by humans of coastal ecosystems is a major threat to the critical services that they provide, valued by Costanza et al. (1997) at US $12.6 trillion. He also noted that changes in coastal water quality and living resources are the result of multiple stressors (Breitburg et al. 1999), and to achieve an integrated viewpoint, any examination of coastal eutrophication needs to consider how anthropogenic nutrient enrichment interacts with other stressors. These include non-indigenous species, habitat loss, fishing, toxic contaminants, freshwater flow manipulation, aquaculture, and climate change (Cloern 2001).

Three inherent physical and biological attributes were suggested that operated in concert to set the sensitivity of individual ecosystems in response to nutrient enrichment (Cloern 2001). The first was tidal energy. A cross-estuary study of 40 individual estuaries found that chlorophyll concentrations in micro-tidal (very small tidal range) estuaries were on average 10 times higher per unit of dissolved inorganic nitrogen (DIN) that in macro-tidal (large tidal range) estuaries (Monbet 1992). The second physical attribute was the set of horizontal transport processes that determine the residence time of water, nutrients, and plankton within coastal basins (Cloern 2001). Algal bloom dynamics are controlled by the balance between the rates of phytoplankton population growth and horizontal transport (Lucas et al. 1999a,b), which in turn are controlled by physical attributes of tide, wind, bathymetry, basin geography, and river flow. Coastal ecosystems with slow transport/long residence times tend to retain exogenous (external origin) nutrients, as they have less efficient 'filter' abilities than coastal systems with short residence times (Nixon et al. 1996). The third attribute was the set of optical properties controlling light exposure to submerged plants, including phytoplankton (Cloern 2001). For some estuarine systems, annual primary production was more strongly correlated with light resource levels than with nutrient resource levels. It was suggested that this helped explain why Chesapeake Bay responded more strongly to nutrient additions than San Francisco Bay, the latter having higher suspended sediment concentrations and higher turbidity (Cloern 1999). The final attribute was the importance of suspension feeders (such as bivalve shellfish) as a biological component of the filter (Cloern 2001). Rates of particle filtering can be high enough to balance the rate of phytoplankton primary production, as a 'top-down' control process (grazing), and can be the key biological component of the filter.

For Danish estuaries, the best predictor of chlorophyll *a* concentration was mussel biomass, rather than measures related to nutrient fluxes or concentrations (Kaas 1996). The same finding was made across 15 Canadian estuaries (Meeuwigs 1999). In Chesapeake Bay, the removal of extensive oyster reefs by overfishing, habitat loss, and disease has reduced the productivity of that system, and contributed to low water quality (Newell 1988, Ulanowicz & Tuttle 1992, Dame & Allen 1996).

4.2.2 Interactions with other stressors

The balance between phytoplankton production and loss to benthic consumers (e.g., filter-feeding bivalves) can be disrupted by the colonisation of coastal ecosystems by non-indigenous species. For example, in northern San Francisco Bay, the Asian clam *Potamocorbula amurensis* became widely established in 1987, and since that time chlorophyll *a* biomass has been persistently low, with primary production being reduced 5-fold (Alpine & Cloern 1992).

Climatically driven pulse inputs of nutrients, from terrestrial run-off (Hama & Handa 1994) or atmospheric deposition (Paerl et al. 1990), can trigger responses such as algal blooms and anoxia. In direct contrast, eutrophication effects are weakest during climatic anomalies of low precipitation and nutrient run-off (Rask et al. 1999). Climatic events change all the physical components of the eutrophication filter: residence time is prolonged during low-flow conditions, and algal blooms frequently develop within estuaries (Relexans et al. 1988), and storm-related events of high river flow can establish strong vertical salinity gradients and development of hypoxia/anoxia in bottom waters (Paerl et al. 1988). In weakly tidal systems, where wind stress is the important mechanism of vertical mixing, weather anomalies can also induce responses. Half the mussel population in the Danish Limfjord died in 1977 following a seven week period of calm winds and warm temperatures, which established a persistent thermal stratification and associated bottom-water anoxia (Mohlenburg 1999).

Climate-driven changes in coastal hydrology and circulation can also affect whether nutrient enrichment effects become significant (Cloern 2001, Levins et al. 2001). For instance, a series of large-scale red tides and regional fish kills occurred in Hong Kong coastal waters during the 1987–88 El Nino. This was presumed to have been a result of changes in the south China Coastal Current, which prevented the offshore transport of *Gyrodinium aureolum* blooms sustained by land-derived nutrients (Yin et al. 1999).

Reversal of human-induced enrichment is possible. In the 1970s, Tampa Bay in the USA showed classic symptoms of over-fertilisation, including high chlorophyll biomass, high turbidity, toxic blooms of the cyanobacterium *Schizothrix calicola*, and the disappearance of vascular plants. Remedial actions included more efficient treatment of municipal waste, and a reduction in phosphorus loading from phosphate-fertiliser producers in the water-shed. By 1980 the annual wastewater loading of N was reduced 10-fold, and within a decade the mean chlorophyll biomass had more than halved, mean secchi depth doubled, the intensity of *S. calicola* blooms greatly reduced, and the seagrass *Halodule wrightsii* began to colonise parts of the bay (Johansson & Lewis 1992).

4.2.3 Eutrophication cascades into fisheries

Impacts on fisheries from eutrophication are not fully understood, with Caddy (2000) commenting that "*synchronous anthropogenic effects on marine coastal systems, particularly since World War II, make it difficult to separate effects of fishing from terrestrial inputs, especially those caused by nutrient run-off*". In fact, at least initially, increases in nutrient loading and associated primary production may result in increases in fisheries productivity (Kerr & Ryder 1992), with Caddy (1993) suggesting that fishery production in formerly oligotrophic seas had increased in recent decades following moderate enrichment from the land.

Kerr & Ryder (1992) recognised four categories of enrichment effects on coastal fisheries: a) modifications to the fish production environment through reduction of suitable habitats for spawning and larval survival, and increased vulnerability, b) changes in fish production habitat, c) change in species/communities due to introductions or replacement of oligotrophic species by those adapted to hypoxia, and d) associated fishery effects of eutrophication and the presence of contaminants (e.g., linkage of flatfish papillomas with eutrophic conditions; Stich et al. (1976)).

Caddy (2000) noted that categories a–c could be easily confused with incidental effects of fishing, such as trawling on muddy sediments adjacent to seagrass beds suspending fine sediments, which in turn reduced the euphotic zone and caused damage to deeper beds. Alternatively, such effects might truly be the result of anthropogenic eutrophication. Caddy (2000) wondered how to distinguish the two effects, or whether they should simply be regarded as synergistic stresses to the ecosystem, following the suggestion of Rapport et al. (1985). He commented that during the 20th century, almost all anthropogenic signals trended in a similar direction, namely towards increased stress on natural freshwater and inshore systems as well as on semi-enclosed marine ecosystems (Rapport et al. 1985, Caddy 1993). Symptoms of this stress include simplifications of ecosystem complexity and dominance by r-selected species.

Caddy (2000) suggested that the catchment basin was the smallest natural unit of landscape, in models that link tightly connected aquatic and terrestrial ecosystems (as summarised by Hornung & Reynolds (1995)). A useful conceptual framework advanced to integrate land-use impacts with those on aquatic ecosystems was that of the marine catchment basin (MCB), defined to include the marine aquatic ecosystem along with the adjacent watersheds that drain into it (Caddy 1993, Caddy & Bakun 1994).

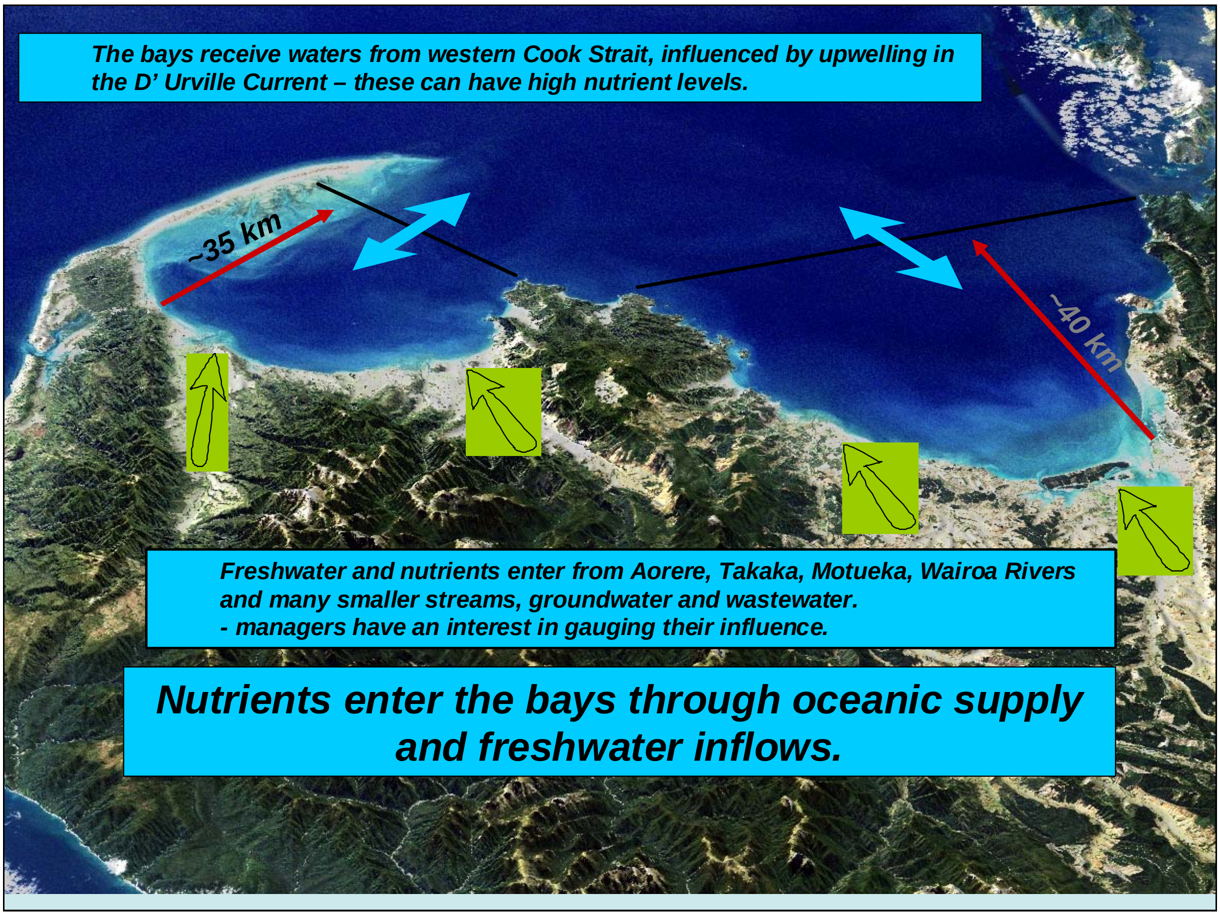
Examples of changes in fisheries production following enrichment include the Mediterranean Sea (Caddy 2000). Until the 1970s, fishery production per shelf area in this region was well below the world average (Gulland 1971). Since then, fisheries productivity has increased over time, especially in the northern region where the rivers Rhone, Po, and Ebro enter the sea, and for the Aegean, where inflows of enriched Black Sea and Marmara waters occur. Contributing factors in the northern areas include increased river run-off, denser human populations, tourism, and intensive agriculture. In direct contrast, along the arid southern and eastern shelves, low production areas still occur under oligotrophic conditions, amplified by the construction of the Aswan barrage (Caddy et al. 1995, Caddy 2000). Following construction of the Aswan Dam on the Nile River, nutrient inputs to the eastern Mediterranean Sea declined by roughly an order of magnitude, and were accompanied by a significant decline in sardine landings. Recovery of fishery production in recent years is associated with increased inputs from large cities and other sources of nutrients draining from the Nile delta into the Mediterranean Sea (Caddy 2000).

In the Black Sea, Sorokin (1994) noted that nutrient discharge onto the shallow northwest shelf rose by an order of magnitude between 1965 and 1974–75, causing phytoplankton blooms and increased turbidity, which adversely affected extensive macrophyte (*Phyllophora*) beds by severely reducing the euphotic zone. These algae formerly contributed dissolved oxygen to near bottom shelf water. Anoxia and growing hydrogen sulphide levels in turn made the extensive mussel (*Mytilus*) beds which were the main bio-filtering agency on the shelf collapse, and led to a 'toxic-shock' effect to the sea as a whole. Zaitsev (1993) observed that despite the overriding influence of environmental change, trawling and harvesting of seaweeds and mussels also contributed to this process: with fines silting over benthos, and further reductions of water transparency which seriously affected light penetration to *Phyllophora* beds, in turn leading to the release of hydrogen sulphide from anoxic sediments.

4.2.4 Eutrophication in New Zealand

There appears to be very little (if any) work in New Zealand on how eutrophication impacts on coastal fisheries. The closest is the work of Savage (2008), who looked at how land-use in catchments affected the adjacent near-shore environment in Southland. Using stable isotopes, a strong positive linear relationship was shown between the percentage of agricultural land in a catchment and the total nitrogen (N) loading to near-shore environments. Sampling of *Ulva* spp. for $\delta^{15}N$ found a clear positive relationship with wastewater nitrogen loads, and that they were a good indicator of land-based nutrients around urban watersheds. This signal was strongest during the maximal seasonal period of growth (spring/summer). Sampling of primary consumer signatures (filtering feeding bivalves – cockles) found no clear relationship with the different terrestrial nutrient sources across estuaries, but did find clear spatial gradients along sites within individual estuaries, which were decoupled from the isotopic gradients of primary producers. This was taken to suggest that there were differences in the dissolved and particulate nitrogen source pools. Sampling of secondary producers (fish: spotties *Notolabrus celidotus*, and estuarine triplefins *Grahamina nigripenne*) found the effects of nutrient enrichment to be transmitted up the food-web, with growth generally enhanced in nutrient-enriched coastal areas. Both species consumed a wider diversity of prey items at pristine sites than in impacted areas. Food-web models suggested that shifts in the relative importance of the different organic matter sources were occurring among the different coastal ecosystems due to nutrient enrichment from land-based activities. While not conclusive, gut content and stable isotope analyses also implied the importance of seagrass production in pristine coastal ecosystems, and suspended particulate organic material (SPOM) (composed largely of phytoplankton) or microphytobenthos in nutrient enriched areas (Savage 2009).

Further north, Zeldis (2008) examined the origin and processing of nutrients in Golden and Tasman Bays (Figure 13). Two nutrient input sources were possible – nutrients entering the bay through freshwater flows, and from the ocean. The two bays are exposed to the oceanic waters of western Cook Strait, which are influenced by upwelling on the west coast, and in general have high nutrient loads. The nutrient climate is also affected by the inputs of four large rivers (the Aorere and Takaka rivers in Golden Bay, and the Motueka and Wairoa rivers in Tasman Bay), as well as many smaller rivers and streams. Mean water residence times of the two bays are 11 and 41 days respectively, with Golden Bay having a smaller volume, a higher net residual freshwater flow, and probably more intense tidal mixing (Zeldis 2008). Measures of the flux of dissolved inorganic nitrogen in Golden Bay found a flux of about 12% of the total to come from rivers, with the remainder deriving from the shelf. Tasman Bay river input was 9%. This finding showed that rather than strongly affecting nutrient supply, the principal role of the freshwater entering the Nelson Bays may be driving their estuarine circulation and in affecting density stratification and turbidity, and in doing so helping drive the local light and nutrient availability for primary producers. Zeldis (2008) suggested that this freshwater influence deserved further research.



The bays receive waters from western Cook Strait, influenced by upwelling in the D' Urville Current – these can have high nutrient levels.

Freshwater and nutrients enter from Aorere, Takaka, Motueka, Wairoa Rivers and many smaller streams, groundwater and wastewater.
- managers have an interest in gauging their influence.

Nutrients enter the bays through oceanic supply and freshwater inflows.

Figure 13: Study area of Tasman and Golden Bays. (Source: John Zeldis, NIWA.)

This system contrasted strongly with that of the Firth of Thames, where on average the riverine supply of organic and inorganic nitrogen to the Firth was greater than the supply coming from mixing across the boundary between the Firth and the Hauraki Gulf (Figure 14) (Zeldis unpublished results). During periods of down-welling domination over the adjacent continental shelf, rivers contributed about 70% of the dissolved inorganic nitrogen (DIN) load, dropping to about 50% when upwelling was active. This present day water quality is significantly enriched, and it is likely that its productivity is substantially higher now than before the arrival of humans. The system is highly ‘net-heterotrophic’, consuming substantial organic matter and producing inorganic nutrients and dissolved inorganic carbon (DIC). It was suggested that pastoral catchment development has resulted in strong effects on the Firth of Thames ecosystem, and that in turn the Firth will respond to changes in catchment management to the extent that it affects nutrient loading (Zeldis unpublished results).

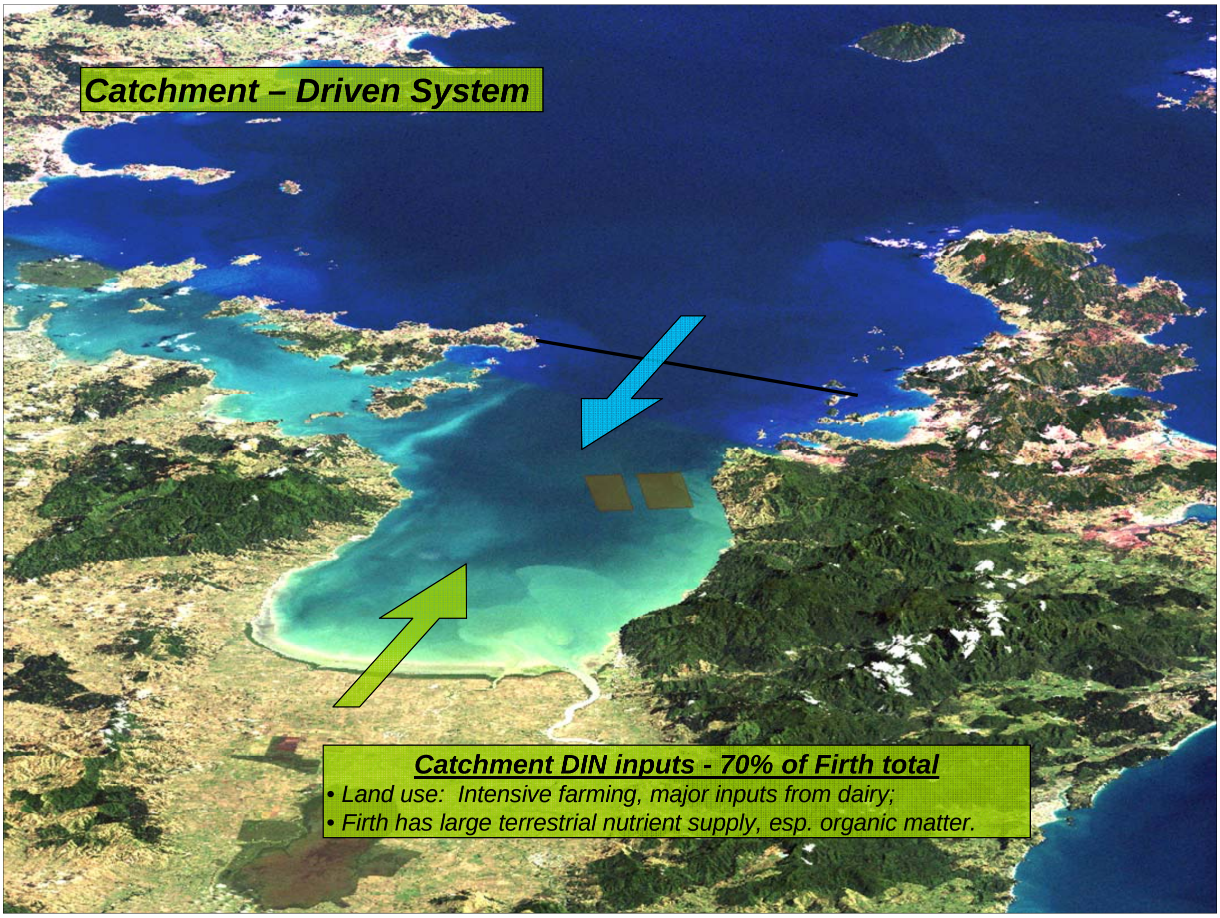


Figure 14: Study area of Firth of Thames. (Source: John Zeldis, NIWA.)

4.3 Effects of freshwater flows and river plumes

River flow is a prominent source of natural variability in coastal ecosystems, which have now become highly manipulated by man. Nearly 80% of the freshwater discharged by large rivers in the northern temperate zone is "*strongly or moderately affected by fragmentation of the river channels by dams and by water regulation resulting from reservoir operation, inter-basin diversion, and irrigation*" (Dynesius & Nilsson 1994). Freshwater is now becoming a scarce resource, with two-thirds of all extractions being used for irrigation. Reduced flows into estuarine environments can increase the salinity of the water column allowing marine flora and fauna to colonise upstream, replacing brackish communities (Wortmann et al. 1997). Alternatively, the opening of floodgates can change salinity to freshwater and back over short time frames, while changes in freshwater flow volumes into coastal marine waters may change temperature and nutrient regimes, alter the extent of estuarine plumes (Grimes & Kingsford 1996), reduce the extent of wetlands, degrade estuarine and nearshore habitat (Serafy et al. 1997), and remove cues for migration (Gillanders & Kingsford 2002).

A number of studies have reviewed the evidence for linkages between the coastal fisheries production of commercially valued crustacean and finfish species, and river flow and plume dynamics. Fisheries production (measured as catch), is often elevated during, or as a consequence of, years with higher water flow, but the causes often remain unproven (Robins et al. 2005). Possible mechanisms that have been advanced include (Robins et al.2005): 1) trophic linkages via changes to primary or secondary production from addition of nutrients 2) changes in distribution as a consequence of altered salinity wedges (expanded, reduced, or connected) 3) changes in population dynamics such as recruitment, growth, survival, and abundance (Drinkwater & Frank 1994, Loneragan & Bunn 1999, Gillanders & Kingsford 2002).

However, there may be a number of steps between the immediate direct effects on physical parameters from changing freshwater flows and the response by estuarine fishery species (Robins 2005). This makes the identification of the causal mechanisms a difficult task. Issues to consider when looking at correlations between changing environmental variables and fisheries yields include, 1) the confounding effects of stock size and fishing pressure (Walters & Collie 1988), 2) the likely non-linearity of linking mechanisms (Baumann 1998), and the probability of multiple mechanisms, 3) the possibility of Type 1 errors (i.e., false significant correlations; Potter et al. 2001), 4) lack of ability to prove causality (Quifoñes & Montes 2001) and 5) their uncertain predictive capability as a consequence of long-term climatic variation or human-induced changes (e.g. habitat loss, pollution). Robins et al. (2005) stressed the need for experimental approaches to address these issues, but also noted that there were significant scale issues to overcome.

In subtropical Australia, significant positive correlations have been found between catch and freshwater flow for mullet (*Mugil* spp.) and flathead (*Platycephalus* spp.) (Lonneragan & Bunn 1999). Conversely, in Chile significant negative correlations between catch and freshwater flow have been found for robalo (*Eleginops maclovinus*) (Quifoñes & Montes 2001). In the United States, catches of red drum (*Scianeops ocellatus*), black drum (*Pogonias cromis*), and spotted seatrout (*Synoscion nebulous*) have been both negatively and positively related to freshwater flows aggregated into two-monthly flows (Powell et al. 2002).

Work on the Rhone River, which flows into the Gulf of Lions (France), found this large river to drive 50% of primary productivity in the Gulf, with inputs of 1–23 million tons of terrestrial matter a year (Salen-Picard et al. 2002). In the Rhone delta polychaetes dominate benthic assemblages, with strong positive temporal fluctuations in response to flooding events, following time lags dependent on species. Opportunistic, short-lived species (e.g., *Mediomastus* sp., *Aricidea claudiae*) showed high, short-term peaks in density and biomass a few months after flooding events, while long-lived species (e.g., *Laonice cirrata*, *Sternaspis scutata*) responded with density and biomass increases with time-lags of 1–3 years, with population increases lasting for several years. In turn, these species supported important finfish fisheries. The common sole (*Solea solea*), with a diet dominated by polychaetes (80%), showed a positive correlation between mean annual discharge and annual commercial landings, with a 5 year time lag, for each of two fishing harbours located close to the Rhone delta (Salen-Picard et al. 2002). It was thought that the long term increase in food after flooding events might favour various life stages of the sole, thus enhancing its population size for several years. Climate had the potential to affect such relationships, as the flow of the Rhone River is related to the North Atlantic Oscillation which drives precipitation (rain) over Western Europe.

Darnaude (2005) followed on with work tracking terrestrial inputs into the food chain of five species of flatfish: scaldfish (*Arnoglossus laterna*), solenette (*Buglossidium luteum*), Atlantic spotted flounder (*Citharis linguatula*), sand sole (*Pegusa* (*solea*) *lascaris*), and common sole and their benthic prey, using stable isotope analyses. Trawl surveys over the 1–100 m depth range showed scaldfish, solenette, and common sole juveniles to be concentrated in the 0–20 m depth zone (98%), while adults occurred at all depths, but with the majority (47–62%) in the 30–50 m depth zone. For the other two species, juveniles and adults had similar distributions, with sand sole sampled from near-shore sandy bottoms (0–20 m), while Atlantic spotted flounder was only found offshore on the muddy bottoms of 30–100 m. Stable isotope signatures of the different fish species were well linked to their benthos diet, and in turn the relative use of terrestrial particulate organic matter (POM) use by the benthos.

At 0–20 m water depth, only deposit-feeding and carnivorous polychaetes exploited terrestrial POM for growth. This was reflected by juveniles and adults of scaldfish, juveniles of solenette, and adults of sand sole (all mainly bivalve and crustacean feeders) being placed at the top of the marine phytoplankton based food web (i.e., these species were relying on marine rather than terrestrial derived material). In contrast, common sole juveniles ingested significant volumes of deposit feeding polychaetes, with stable isotope analyses showing them to depend heavily on terrestrial POM for their growth (in agreement with the findings of Salen-Picard et al. (2002)). Adult common sole, solenette, and juvenile sand sole (deposit feeding or carnivorous polychaete feeders) had intermediate isotopic positions, indicating reliance on marine primary production, but with some exploitation of terrestrial POM.

At 30–50 m water depth, terrestrial POM was exploited by juvenile bivalves, brachyurans (crabs), and shrimps (Crangonidae), and all polychaetes. Atlantic spotted flounder seldom consumed polychaetes, and was positioned at the top of the marine POM-based food web, with some occasional terrestrial influences from the consumption of shrimps and polychaetes. Adults of scaldfish, solenette, and common sole had isotopic signatures fully explained by the benthic prey consumed, with higher levels of terrestrial POM contributions.

At 70–100 m, benthic organisms made little use of river POM, with only subsurface deposit-feeding polychaetes reliant on terrestrial matter for growth. Atlantic spotted flounder juveniles and adults did not exploit these, and there was also very low use by adult scaldfish and solenette, which placed these fish at the top of the marine POM-based food web. Conversely, adult common sole did ingest substantial volumes of these polychaetes, and were positioned between the marine and terrestrial POM-based food webs.

Overall, Darnaude (2005) concluded that terrestrial POM contributions were least in Atlantic spotted flounder, whose diet always included very little prey that used terrestrial POM. It was greatest in common sole, where consumption of deposit-feeding polychaetes by juvenile and adult fish was high, and where adults were mainly found at 30–50 m water depths. It was intermediate for the remaining three species (solenette, scaldfish, and sand sole), with the highest contributions occurring in those life stages where the ingestion of polychaetes was highest, and/or where life stages were present at 30–50 m water depth. These findings demonstrate that terrestrial inputs into coastal fisheries can be significant, but may vary by species, and have greatest influence at different life cycle stages dependent on the species involved. Darnaude (2005) also commented that this system was a relatively simple one in terms of primary producer signatures (e.g., marine vegetation was absent) and that it was also possible that non-negligible uptake of terrestrial POM by marine benthic fish might be occurring in other areas, where system complexity has prevented its detection by stable isotope methods.

A further possible mechanism that was suggested was the impact of variation in inter-annual terrestrial POM inputs on fish reproductive success. Feeding success affects both size-at-first maturity, and adult growth and condition, which in turn regulate the success of spawning activities. It was suggested that terrestrial POM uptake by adult fish after a flood would optimise species reproductive success for several years, eventually resulting in a long-term increase in fishery catches, with a time lag dependent on spawning frequency and age at recruitment to the fishery. This phenomenon, already shown for common sole by Salen-Picard et al. (2002), was suggested to also occur in solenette, and to a lesser extent, scaldfish and sand sole.

4.3.1 Freshwater flows and river plume effects on New Zealand fisheries

There appears to be little (no) work on the effects of freshwater flows and river plumes on coastal fisheries in New Zealand. We could find no relevant quantitative literature, beyond work done in the Motueka River, which has not yet directly linked the influence of flows to adjacent coastal fisheries (see scallop 5.2.4).

5. WHICH SPECIFIC NEW ZEALAND FISHERIES SPECIES ARE LIKELY TO BE IMPACTED?

5.1 Coastal fisheries

New Zealand's extensive coastline, encompassing more than 1 600 km of latitude for the main islands, supports a diverse and substantial range of coastal invertebrate and finfish fisheries. Pipi (*Paphies australis*) and cockles (tuangi) (*Chione stutchburyi*) are culturally and recreationally harvested throughout the country, especially close to large population centres. The main commercial fishery for pipi is at the Whangarei Harbour entrance, Northland (Mair Bank), while commercial cockle fisheries exist inside Whangarei Harbour (Snake Bay), further south in Tasman/Golden Bays (Pakawau Beach, Tapu Bay-Riwaka, Ferry Point), and in Otago (Papanui, Waitati, Purakanui Inlets and Otago Harbour). Total 2006–07 annual cockle and pipi catches were 1 345, and 135 t respectively (all catches given from Ministry of Fisheries 2008). On exposed beaches, tuatua (*Paphies subtriangulata*, *Paphies donacina*) replace cockles and pipi, with a small commercial fishery (currently not operating) inside the mouth of the Kaipara Harbour, and non-commercial harvesting around the country on exposed beaches, especially in northern New Zealand. Commercial scallop (*Pecten novaezelandiae*) fisheries are focussed around east Northland (242 t), Coromandel (includes greater Hauraki Gulf) (471 t) and the Nelson/Marlborough areas (NB: scallop landings vary strongly from year to year), while small non-commercial fisheries exist in these regions and more widely, including west coast North Island harbours, and Paterson Inlet, Stewart Island. Important reef-based fisheries include kina (*Evechinus chloroticus*) (868 t), paua (*Haliotis iris*) (995 t), and rock lobster (*Jasus edwardsii*), with catch levels varying between different parts of the country, while small localised fisheries for paddle crabs (*Ovalipes catharus*) (172 t) exist off sandy beaches in more exposed areas. Flatfish (flounder and sole) fisheries are important around the country, with different regions being dominated by different species (7 in total, generically recorded in fisheries statistics as FLA – 4 050 t). Inside larger northern estuarine systems (e.g., Manukau and Kaipara), and in shallow coastal environments (e.g., Firth of Thames and Canterbury Bight) there are regionally important flatfish species, including yellow-belly flounder (*Rhombosolea leporina*) in the north, and sand flounder (*Rhombosolea plebeia*) more nationally.

In more northern areas, snapper (*Pagrus auratus*) dominate commercial catches (6 328 t), along with a related fishery for trevally (*Pseudocaranx dentex*) (2 987 t), leatherjackets (*Pakiri scaber*) (454 t), and with kingfish (*Seriola lalandi*) as a non-target bycatch fishery only (161 t). Also in this region, and nationally, important species include red gurnard (*Chelidonichthys kumu*) (3 817 t), John dory (*Zeus faber*) (815 t), kahawai (*Arripis trutta*) (2 500 t), and tarakihi (*Nemadactylus macropterus*) (5 729 t). Further south, on shallow rocky reefs and on biogenic (living) reefs, blue cod (*Parapercis colias*) become important (2 403 t), along with blue moki (*Latridopsis ciliaris*) (504 t), while soft sediment associated species include red cod (*Pseudophycis bachus*) (5 551 t) and elephantfish (*Callorhinchus milii*) (1 148 t). School shark (*Galeorhinus galeus*) (3 719 t) and rig (*Mustelus lenticulatus*) (1 362 t) (a dogfish species) also support regional fisheries. In some regions, semi-pelagic and/or pelagic species are targeted by purse seining, as well as midwater trawls, including kahawai, skipjack tuna (*Katsuwonus pelamis*), blue mackerel (*Scomber australasicus*) (10 688 t), and trevally. In the high energy environment of Foveaux Strait, the iconic Bluff oyster (*Ostrea chilensis*) fishery (Figure 15) still operates (7.37 million oysters harvested 2006–07), albeit at a level much reduced from in past decades, along with a smaller fishery in Nelson/Marlborough (132 t).

Figure 15: Oyster vessels in Bluff Harbour. (Source: Ministry for the Environment.)

Collectively, these cultural, recreational and commercial coastal fisheries generate important social and economic benefits to different regions, and New Zealand as a whole. In addition to fishing intensity, other factors can affect their ongoing health and production. Increasing evidence is indicating that some of the environmental conditions required in maintaining populations (which in return support fisheries) have been significantly adversely affected in the past, and that these environmental stresses and associated effects are continuing to operate in the present day.

5.2 Filter-feeding bivalves

Filter-feeding shellfish are especially vulnerable to sedimentation. Fished species include cockles and pipi on intertidal/shallow subtidal estuarine flats, and in sheltered bays. Tuatua dominate on more exposed surf beaches, along with toheroa at some locations. Subtidally, pipi are found in estuarine subtidal channels with coarser sediments and strong tidal flows, while a number of surf clam species are found along exposed surf beaches around the country. In subtidal estuarine areas, around islands, and in open coastal embayments, scallops often occur, supporting commercial fisheries in Northland, Coromandel, and Golden/Tasman Bays/Marlborough Sounds.

5.2.1 Cockles (*Chione stutchburyi*) and pipi (*Paphies australis*)

Cockles and pipi are ubiquitous species of estuaries and sheltered shores, with non-commercial fisheries at numerous sites around New Zealand. Virtually all commercially harvested pipi come from Mair Bank, at the entrance to Whangarei Harbour, while cockles are collected from Whangarei Harbour (Snake Bank), Golden/Tasman Bays, and in Southland estuaries (Ministry of Fisheries 2008). Despite these species' present-day widespread abundance, there are numerous anecdotal accounts of substantial declines in their distribution and abundance from many places around New Zealand, associated with increasing human-induced sedimentation and environmental stress, and/or strong recreational harvesting pressures. Many estuarine sites that today are mud and silt dominated have dense layers of dead cockle shell starting several feet below the surface, suggesting historical habitat (and associated species) changes. A monitoring time series of intertidal shellfish populations (largely cockles and pipi) exists for a range of beaches in the Auckland region, which began in 1992 following concerns about shellfish declines. Annual surveys have been undertaken since then, terminating in the summer of 2006, but scheduled to recommence in 2009. A series of Ministry of Fisheries reports have documented these surveys (Cook et al. 1994, Pawley et al. 1997, Morrison & Brown 1999, Morrison et al. 1999, Akroyd et al. 2000, Walshe & Akroyd 2002, 2003, 2004, Walshe et al. 2005, 2006, Pawley & Ford 2007), along with a study that looked at three beaches in detail, with respect to their natural shellfish dynamics versus the levels of recreational harvesting being extracted (Hartill et al. 2005). These have been strongly focussed on the estimation of numbers and biomass of the populations over time, and human harvest levels. Little associated temporal environmental information has been recorded, at least within the programme itself. No full 'meta-analysis' has been undertaken on these data, although Grant & Hay (2003) reviewed the series (as of 2002) as part of a larger review on issues related to intertidal shellfish population depletions in the Hauraki Gulf Marine Park. They concluded that "*Overall, the majority of sites surveyed over the various studies show decreasing trends in inter-tidal infaunal bivalve abundance. However, the small quantity of robust data available makes generalisation to the Hauraki Gulf Marine Park as a whole inappropriate. We note that observation of the depletion of infaunal inter-tidal shellfish stocks in the Hauraki Gulf Marine Park region appears to be supported by a substantial body of anecdotal evidence*".

Cockles and pipi have preferences for particular bottom sediment compositions. Anderson (2008) looked at the relationship between maximal numbers of cockles and pipi, and the percentage of mud (particles less than 63 microns – 0.063 mm), across more than 3 000 benthic cores collected from numerous Auckland estuaries. She found clear sediment preference curves, with the optimum mud percentage for pipi being 3.4% (confidence intervals (CI), 3.3, 3.5), while cockles were more tolerant at 11.3% (CI 7.7, 14.8). These findings agree with earlier work by Thrush et al. (2003). Tank-based experiments using a range of suspended sediment concentrations have also found strong effects. Figure 16 (adapted from Schwarz et al. 2006a) gives a summary of the total suspended sediment (TSS) ranges used in New Zealand laboratory based studies, as well as field measurements. Nicholls et al. (2003) found increasing physiological stress with increasing suspended sediment concentrations. Both species continued feeding at high levels of suspended sediment concentrations over the short term (one week), but for periods longer than this, their condition was adversely affected. Different types of sediment had different levels of effect, with terrigenous sediment having a greater effect on cockles than marine sediment (Gibbs & Hewitt 2004). Work in the Whitford Embayment, Auckland (8 sites), found that high suspended sediment concentrations adversely affected juvenile cockle growth rates, while the reproductive status of adult cockles and pipi was also negatively affected (Gibbs & Hewitt 2004).

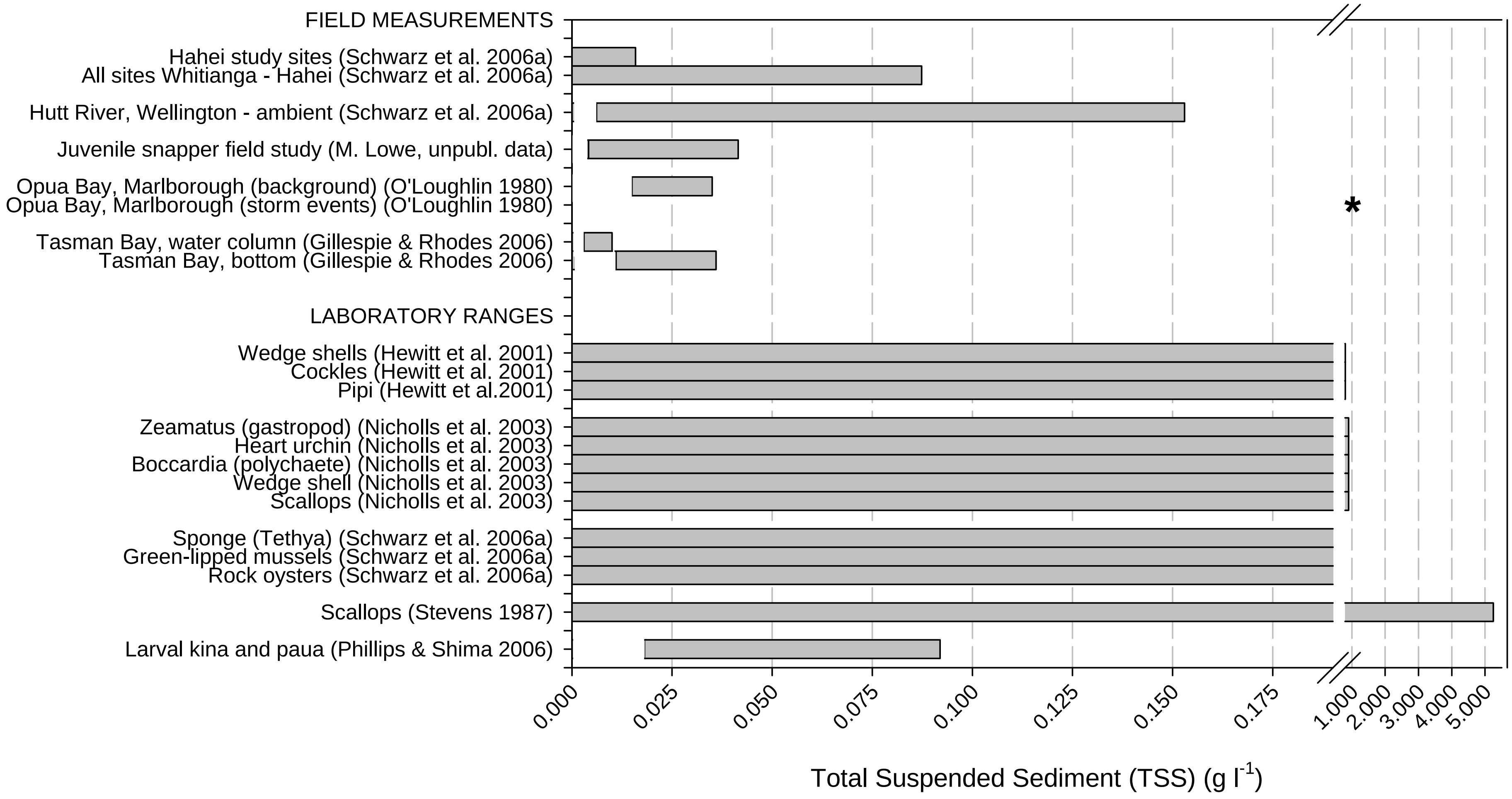


Figure 16: Summary of total suspended sediment concentrations from a range of New Zealand studies discussed in this report (adapted from Schwarz et al. 2006a). *, storm event.

5.2.2 Tuatua (*Paphies subtriangulata, P. donacina*), toheroa (*P. ventricosa*), and surf clams (various species)

On exposed surf beaches, tuatua, and to a much lesser extent, toheroa shellfish populations dominate intertidally. A small commercial fishery for tuatua operates intermittently in the entrance to the Kaipara Harbour, while non-commercial fishing occurs throughout New Zealand. Relatively little is known about this species, apart from some work on distribution and abundance patterns, and reproductive cycles in a northern population of *Paphies subtriangulata* (Grant 1994, Grant & Creese 1995), and population surveys off Brighton Beach in Christchurch (Cranfield et al. 2002) for the more southern species *Paphies donacina*. Possible environmental stressors on this species are unknown.

Toheroa historically once supported regionally important fisheries in west Northland, the Wellington coast, and Southland, but population levels declined strongly following over-fishing in the early to mid 20th century (concerns were voiced as far back as 1926 (Anon 1926)). All toheroa beds are now closed to recreational and commercial fishing (Morrison & Parkinson 2008). In the several decades since fishing was halted (notwithstanding poaching issues) these populations have failed to recover, suggesting that while over-fishing may have driven these populations to extremely low levels, other non-fishing factors may now be acting to prevent populations from recovering. Time series data for northern populations (Ninety Mile, Dargaville, and Muriwai beaches) suggest that these beaches receive erratic (if occasionally quite substantial) recruitment pulses, followed by large-scale mortality that prevents increases in the abundance of large toheroa (Morrison & Parkinson 2008). In general, it appears that some aspect of toheroa population dynamics, or their supporting habitats, has changed so that areas once supporting abundant populations are no longer able to do so. Reasons for these declines are not well understood, but are likely to include a combination of over-harvesting, environmental changes, and other impacts such as heavy vehicle traffic along some beaches (Ninety Mile Beach for instance is officially classed as a state highway). Changing land-use is one of several possibilities. Sub-fossil wood samples indicate that the land adjacent to Ninety Mile Beach was largely covered in coastal broadleaf forest before the arrival of humans (Coster 1983); today there are a number of exotic pine forest plantations, while a large number of coastal lakes and other surface water features once common in the area no longer exist. There is speculation that changes in water run-off to Ninety Mile Beach have negatively influenced toheroa populations; often sites of higher abundance seem to correlate with areas of freshwater seeps from the land, with the suggestion made that pine forests reduce these seeps. A current information synthesis and review of toheroa population dynamics and factors influencing these dynamics may reveal new insights into such potential processes (see Section 10).

Surf clam assemblages (a number of species) also occur off many of New Zealand's beaches (Cranfield & Michael 2001, Taylor & Morrison 2008), and support emerging commercial fisheries in several areas (Triantafillos & Maxwell 2008). Nothing is known about their potential responses to land-based stressors, although they occur in spatial areas likely to receive regular inputs from the adjacent land catchments.

5.2.3 Green-lipped mussels (*Perna canalicus*)

Green-lipped mussels are another widespread shellfish species that has undergone large-scale declines in abundance associated with human activities. Mussels, like many other shellfish that occur in dense beds, are 'ecosystem engineers', meaning that they have a strong influence on the wider functioning of other species and the local ecosystem. For instance, high density populations have large filtering capabilities, which increase links between the seafloor and water column productivity (bentho-pelagic coupling) and may exert strong controls on phytoplankton populations and eutrophication effects (Cloern 2001, see Section 4.2). They also increase the local biodiversity of an area by providing more complex habitats (Ragnarsson & Raffaelli 1999), and probably act as important nursery habitats for juvenile fish (e.g. strong spotty and triplefin associations have been documented (I. McLeod, Leigh Marine Laboratory, unpubl. data)), and as foraging areas for adult fish.

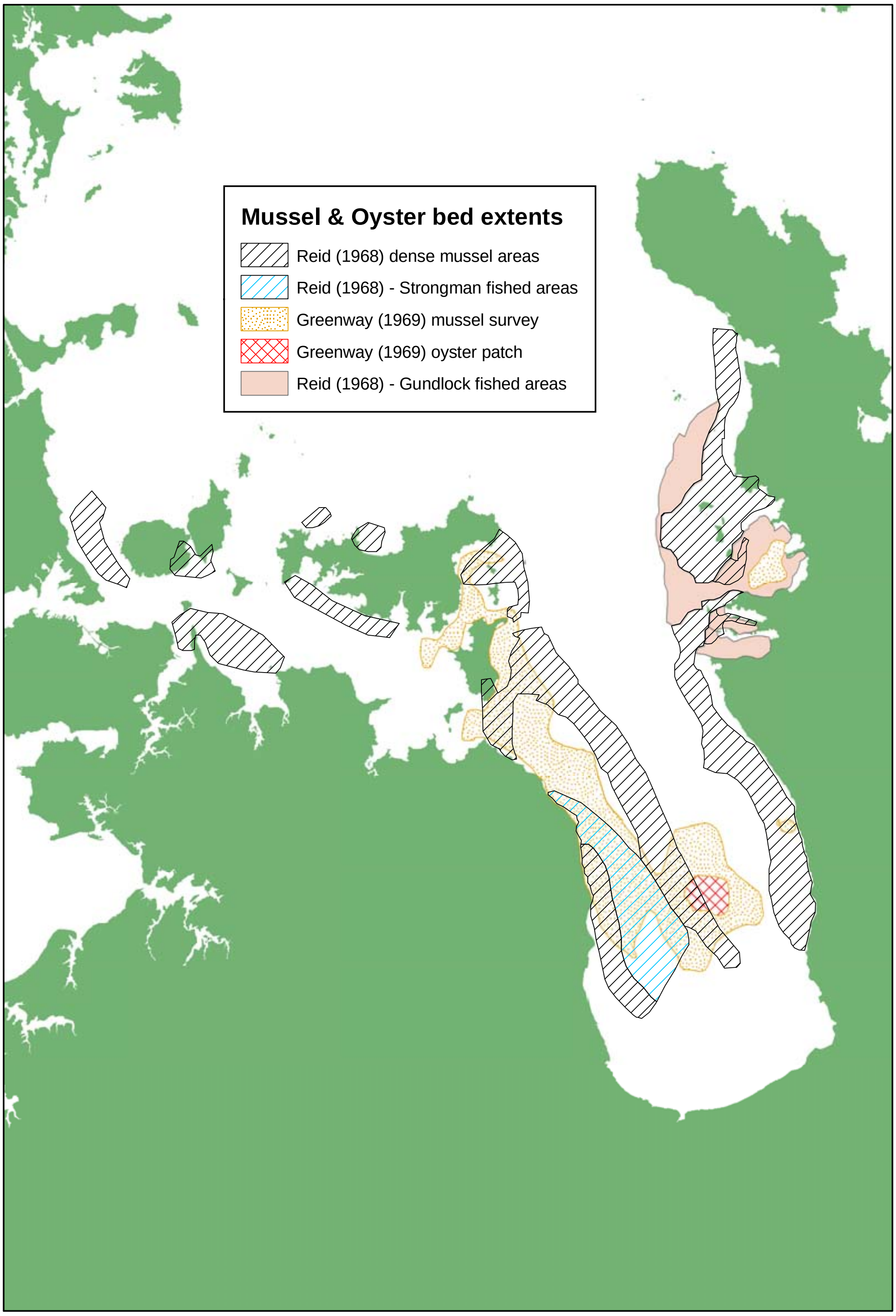


Figure 17: Historical green-lipped mussel distribution in the inner Hauraki Gulf and Firth of Thames. Reid (1968) used interviews with the two families who dominated the fishery to draw maps of where the dense mussel beds once were, as well as additional areas fished by either Strongman or Gundlock boats (which used different types of dredge, and boats with different towing power). Reid (1969) drew a map of mussel distribution based on his 1961 survey (206 stations), as well as identifying a small areas of dredge oysters in the centre of the Firth of Thames. Mismatch of coastline and mussels is due to inaccuracies in the original maps. (Sources: Reid 1968, Greenway 1969.)

From the 1920s to the 1960s, a commercial dredge fishery operated in the Firth of Thames and the inner Hauraki Gulf (Figure 17), until the fishery became uneconomic due to very low population densities (Reid 1968, Greenway 1969).

Surveys in 2002 and 2003, using the acoustic technology QTC IMPACT, able to detect green-lipped mussel beds remotely (Morrison et al., accepted), did not find any evidence of recovery of these beds (Morrison et al. 2002, 2003). Associated ground-truthing with a towed video sled, at sites selected across the different acoustic classes, found only occasional small clumps of mussels at two sites, in association with coarser seafloor sediments – the biggest being about 1 m^2 in extent.

Thus, more than 40 years after the fishery ceased, these mussel populations have not returned. While the definite reasons are not known, it is strongly suspected that the fine sediment nature of the seafloor, with little surface structure (e.g., dead shells, hydroids, red algae) for larvae to settle on, and ongoing silt re-suspension from storms, may prevent successful larval settlement and growth. Extensive mussel farms are now present in the Firth of Thames, and under some of these, shell drops have resulted in the successful establishment of mussel clumps on the seafloor at the hundreds of metres scale (I. McLeod, Leigh Marine Laboratory, unpubl. data). Larval mussel spat supply is not thought to be a problem, with spat being caught by farm operators running experimental spat-collecting lines, and in earlier work in the 1980s in scallop-spat collectors deployed in the area (Bartrom 1990). As with other species such as scallops, green-lipped larvae require foliose settlement surface to settle on when making the life stage shift from the water column to the seafloor. For example, the collection of mussel spat for the aquaculture industry is strongly dependent on beach-cast supplies from Ninety Mile Beach, where mussel spat are washed up attached to drift material, primarily algae and hydroids, with more than 70 t being harvested each year (Alfaro & Jeffs 2002). Most of these are red algae from the subtidal, dominated by *Osmundaria colensoi*, *Carpophyllum augustifolium*, and *Rhodymenia dichtotoma* (Alfaro & Jeffs 2002). Such 'nursery' species are adversely affected by high silt loads in their own right, e.g., hydroids are filter-feeders, while algae require sufficient light levels to photosynthesise. Their loss from the system may help create a population 'bottleneck' for larval settlement and/or juvenile phases, resulting in low (or no) adult population abundances. An experiment to look at these different possibilities is under way on the western side of the Firth of Thames (I. Mcleod – see Section 10).

New Zealand is not alone in having lost significant areas of mussel beds. For instance, there have been significant declines in the extent of wild intertidal mussel (*Mytilus edulis*) beds reported from large coastal areas off Germany, the Netherlands, and Denmark, so that they are now rare in the Wadden Sea (OSPAR Commission, Wolff 2005, in Lotze et al. 2006) and considered to be threatened in the United Kingdom (Hiscock et al. 2005).

5.2.4 Scallops (*Pecten noveazelandiae*)

Scallops are an important and highly valued coastal shellfish species. They tend to occur in aggregations (beds) often associated with coarser sea-floor sediments, and reasonable current flow. Most monitoring of scallop populations has revolved around annual assessments of commercially fished populations, to assist in the setting of yields for the year under the premise of Current Annual Yields (CAY) – a concept created for managing stocks that can fluctuate strongly in their overall abundance from year to year – scallops being a classic example of such behaviour. No work has been specifically directed at the relationships between scallop bed occurrence and environmental factors. However, anecdotal evidence suggests the loss of some beds close to shore that are adjacent to significant terrestrial run-off. Nesbit (1998) commented on observations of large scale mortalities of scallops in the Kaipara Harbour, associated with periods of heavy rainfalls and extensive brown discoloration of harbour waters (scallop beds of that harbour are currently under a rahui due to perceived low numbers). Historically, abundant scallop beds in Whangarei Harbour were associated with subtidal seagrass meadows, which disappeared in the 1960s along with the seagrass – see Section 5.5.1 – Morrison (2003). Casual observations of the inner Kawau Bay area, in particular around Rabbit and Terakihi Islands, suggest that scallop beds were once common around the islands as evidenced by dead shell (which can last for very long time periods on the seafloor), but living scallops are now no longer observed (M. Morrison pers. obs.). The adjacent Matakana River discharges high volumes of suspended sediments during storm events, as evidenced by large discoloured plumes visible from the shore, and an associated fine silt seafloor bottom occurring as a large ‘foot-print’ around the estuary mouth area and beyond (M. Morrison pers. obs.). As with green-lipped mussels and many other bivalves, scallops require foliose settlement surfaces to settle onto as larvae from the water column. This is why artificial scallop spat collecting bags, consisting of fine netting, work so well in scallop enhancement operations. Such foliose surfaces are adversely affected by high levels of sedimentation. We have no information on what these settlement surfaces are likely to be New Zealand, but given overseas knowledge, and the wide geographical range of scallops in New Zealand, they are likely to be generic rather than specific species (e.g., hydroids, algae, and emergent shell).

Scallops have been shown to be sensitive to suspended sediment concentrations under laboratory conditions. Lab-based work by Stevens (1987) using excised scallop gill tissue as an assay device, found decreasing crawl velocities (gill tissue has many cilia) in response to increasing silt concentrations (0.025, 0.05, 0.1, 0.2, 0.3, 0.5% silt by dry weight), and silt size (less than 10 microns, 10–30 microns). It was suggested that crawl speeds changed due to reductions in dissolved oxygen, with increasing turbidity levels (silts) depleting oxygen levels. Concurrent tank-based experiments with whole scallops (20 mm size) found increasing behavioural stress and associated mortalities in response to increasing silt concentration and decreasing particle size. In the first 15 minutes, scallops rapidly drew in and expelled water, or swam around within their tray. After 30 minutes, they settled with open valves, and after 2–3 hours silt covered the ctendaria (eyes), silt was in the mantle cavity, and the mantle tentacles were withdrawn. By 5–6 hours the mantle edge’s response to mechanical stimulation was weak. Dissolved oxygen levels in the tanks declined from 6.5–7 down to 1–1.5 ppm after 15 hours. Work by Nicholson (1978) showed similar trends, with scallops reducing their pumping rates by 92% in response to silt concentrations of 80 mg l^{-1} (0.8% dry weight). Yamamoto (1957) found that suspended mud led to a blocking up with fine-grained particles of scallop gills (*Patinopecten* (*Mizuhopecten*) *yessoesnsis*), preventing normal respiration. Small scallops are known to have higher oxygen demands per unit body weight of tissue, with an inability to utilise internal supplies within body fluids. They also cannot respire anaerobically through the metabolism of the crystalline style (one of the organs used for feeding) (Dugal 1939, Yamamoto 1957), making them more susceptible than adults to adverse environmental conditions. More recent work by Nicholls et al. (2003) found that scallops were able to feed at high levels of suspended sediments over short time intervals (one week) but that their condition was adversely affected by high concentrations over longer time periods. Concentrations over 100 mg l^{-1} were correlated with increasing variability in clearance rates, suggesting adverse effects on scallops’ abilities to process the suspended particles (Nicholls et al. 2003).

Bottom type may also affect the growth and survival of scallops. Most scallop beds tend to occur over coarser seafloor sediments such as sands, shell hash, and shell grit, and are far less common over silt and mud. Silina & Zhukova (2007) assessed two neighbouring populations of the Japanese sea scallop *Patinopecten* (*Mizuhopecten*) *yessoesnsis*, which experienced similar hydrological regimes (water temperature, salinity, currents) but with different bottom types – sand, and muddy silt. While food availability was higher at the muddy site, and scallops appeared more food limited at the sandy site, scallops actually grew significantly faster at the sandy site. Measurement of near-bottom water oxygen concentrations found these to be much lower at the muddy site. It was suggested that this factor, in combination with the high resuspension of inorganic fine-grained particles enriched with inedible dead organic matter (material not useful for scallops, but taking energy to filter out from edible material), was the main reason for the reductions in growth of scallops relative to the sandy site. Similar examples of differences in growth and condition indexes between sand and mud bottom types have been quantified for the clams *Rangia cuneata* (Peddicord 1977), *Mya aernaria*, and *Mercenaria mercenaria* (Newell & Hidu 1982, Grizzle & Morin 1989), and the oyster *Ostrea edulis* (Grant et al. 1990).

A possible New Zealand example of such effects on scallops comes from work by Gillespie & Rhodes (2006). They assessed the quantity and quality of near-bottom and water column suspended particulate material (SPM) at an inshore site in Tasman Bay, strongly influenced by the Motueka River plume (sampled 24–25 November 1998 and 23–25 February 1999). Major short-term fluctuations in turbidity, referred to as the Near Bottom High Turbidity (NBHT) layer, were found up to 0.5 m above the seafloor, and attributed to the river plume. Sampling of SPM in this layer (at 50 mm above the seafloor) found it to be largely composed of inorganic sediments, and therefore of poor nutritional quality for benthic suspension feeding bivalves. Sampling higher up in the water column (over 0.5 metres) found higher proportions of microalgae and/or other organic materials. Comparison of feeding activity of scallops on the seafloor versus ones suspended in the water column found those on the seafloor to temporarily stop feeding while in the high turbidity layer, while those in the water column fed on uninterrupted. The origin of the fine material in the NBHT is thought to be from flood-related discharges coming from the Motueka River catchment, with its long-term persistence being due to recurring tidal re-suspension of fines. While the NBHT is a natural feature of the plume, it was thought to be have been considerably exacerbated into a chronic long-term phenomena by repeated physical disturbances of the seabed by human activities such as dredging and trawling, which compromise the integrity of the sediment-water interface (Gillespie & Rhodes 2006). Scallop harvests in Tasman Bay are now at their lowest level since catch records began, and it was suggested that the quantity and quality of SPM available for benthic suspension feeders may be a major contributing factor (Gillespie & Rhodes 2006). A desktop-based review has been commissioned by the Ministry of Fisheries from NIWA, and is now in progress to assess possible contributing factors to these very low Tasman Bay stock abundances, including factors such as changing land-use in the Tasman catchment, human impacts, or climatic and marine environmental factors.

5.2.5 Paua (*Haliotis iris*) and kina (*Evechinus chloroticus*)

Paua and kina are highly valued, with the first supporting very important commercial fisheries, as well as customary and recreational take. Little quantitative information appears to exist on how land-based effects directly impact on adult populations, althrough gross alteration of some inshore reef habitats by sedimentation seems to be having strong negative effects on local fisheries (e.g., Mahia Peninsula, Miller et al. 2007). Some work has been undertaken on the larval and settlement phases of these shellfish. Phillips & Shima (2006) looked at the effects of suspended sediments on kina and paua larvae, across six concentration levels (including a control). They found that stage specific (different larval forms) mortality rates of urchins increased with suspended sediment concentrations, but decreased with age. Paua showed similar patterns, but with older larvae also continuing to experience high losses when exposed to sediments.

Mortality rates of both species increased in response to acute exposure to sediments early in their development. Kina mortality was immediate, coinciding only with exposure to sediments, and ceasing once the stressor was removed. Paua mortality persisted well after the removal of sediments, making them overall much more vulnerable to cumulative effects. Overall, cumulative survival to competency was similar for the two species, and generally decreased with increasing suspended sediment exposure time and/or concentrations. Phillips and Shima noted that natural cohorts of paua larvae were probably more likely to encounter plumes of suspended sediments transiently, rather than through their entire larval period, but that even this transient exposure would result in substantial mortality. Other synergistic stressors, such as reduced salinity and increased toxins, are also likely to be a component of such plumes, and may play important roles.

Work by Walker (2007) on kina found wave exposed reefs to have higher densities (1.7–9.6 per 100 cm^{-2}) than wave-sheltered reefs (0.1–6.6 per 100 cm^{-2}), with juvenile urchins (under 30 mm) being 25 times more abundant on wave exposed reefs. Newly settled urchin recruits (2–5 mm) densities on wave-exposed reefs ranged from 0.2 to 0.6 per 100 cm^{-2}, and were undetectable on sheltered reefs. Laboratory experiments showed that fine sediment concentrations of one- to two-thirds the level found in the field at wave-sheltered reefs inhibited larval kina settlement, and reduced the survival of both recruits and juvenile kina. It was suggested that sedimentation may be adversely affecting their population dynamics at the sheltered reef locations. Similar results have been documented for the red abalone, *Haliotis refescens* (Raimondi et al. 1997).

5.2.6 Rock lobsters (*Jasus edwardsii*)

We could find no quantitative information on the potential impacts of land-based activities on rock lobster populations in New Zealand. However, anecdotal accounts of reduced lobster growth rates in CRA 2 (east coast North Island) have been discussed in Ministry of Fisheries Working Groups (K. Sullivan, Ministry of Fisheries, pers. comm.), along with the possibility of reduced growth rates for scampi (*Metanephrops challengeri*) in deeper waters off the shelf relative to other areas. The possibility of a sedimentation effect on growth rates has been raised. Work on a related lobster species (*Jasus lalandii*) in the Benguela ecosystem (South Africa and Namibia) has found a strong negative correlation between rock lobster catches and major flood events from the Orange River (Penney et al. 2007). This fishery has existed since the late 19th century, and was for many years the world's largest fishery for a *Jasus* species. South African catches have since fallen from a peak of 16 572 tons in 1951, to about 2 300 t in recent years (an 86% decline), while Namibian catches have fallen from a peak of 9 189 t in 1955 to about 290 t in recent years (a 92% decline) (Penney et al. 2007). While Orange River floods have declined from historical times following the construction of numerous dams, in 1988 the heaviest flood on record discharged 24.3 km^3 of water, and 80.9 million tons of sediment (almost five times the average annual sediment input) over a three month period. It has been postulated that the sedimentation deposition from this flood may have been a major contributor to the subsequent declines in growth rates and productivity of both the South African and Namibian rock lobster fisheries, through smothering of near-shore food resources (Penney et al. 2007). Freshwater effects were probably also important, with impacts evident for up to at least 140 km south of the river mouth, almost completely eliminating benthic organisms between 5 m depth and the mid-littoral zone (Branch et al. 1990, in Penney et al. 2007). Given these probable flood effects, and that New Zealand contributes almost 1% of the world's sediment load from rivers, similar (albeit smaller spatial scale) impacts may also be occurring in New Zealand's coastal environment, especially in regions such as Hawkes Bay.

5.2.7 Other species on the open coast

Our understanding of the impacts of land-based activities on open coast species is modest. Of note are the observations of McKnight (1969) of a site off the east coast of Coromandel, in 22 m of water near Kennedy Bay. While sampling with a surface-operated grab, a site was encountered consisting of a compact shelly sand layer, overlain by a mud layer composed entirely of mud/silt. This mud was described as semi-fluid and brown, with grey-black streaks. Individuals of the filter-feeding bivalves *Nucula nitidula*, *Dosinea subrosea*, *Scalpomactra scalpellum*, and fragments of *Longimactra elongata* were found in the sand layer, along with the dominant species *Tawera spissa*, individuals of which were noted to still have bits of tissue attached to the shells. The last heavy rainfall event had been 24 days earlier. This sample was interpreted to be evidence of a catastrophic burial of a bivalve assemblage (McKnight 1969).

A marine core taken from east of Poverty Bay, and analysed for sediment and pollen content, showed that sedimentation rates increased by an order of magnitude following European conversion of native scrub and forest into pasture (Wilmshurst et al. 1997) (although the core location was susceptible to some sediment disturbance). Foster & Carter (1997) concluded that the Holocene (10 000 years ago – present) sedimentation rate on the continental shelf was almost five times less than the rate measured since deforestation by Europeans of the eastern Hawke's Bay region, Gisborne, and East Cape. It is within the realms of possibility that many of the inshore benthic communities that we know and recognise in finer sediment areas (e.g., the heart urchin and brittle-star assemblages of the inner Hauraki Gulf (Powell 1937) and Marlborough Sounds (Handley 2006), are fundamentally different from what existed before the adjacent catchments were cleared of forest.

5.3 Finfish

In New Zealand, evidence is emerging of sub-lethal concentrations of suspended sediments affecting the upstream migrations of native freshwater fish species, of which 70% are diadromous (McDowall 1990). Feeding experiments on migrant banded kokopu (*Galaxis fasciatus*) showed reduced feeding rates above 25 NTU, increased avoidance reactions, and decreased migration rate in natural streams (Boubée et al. 1997, Rowe & Dean 1998). Field studies suggest that turbidity occurs over this level during the whitebait (juvenile galaxids) migration season (August–December), for more than 10% of the time, which could reduce the upstream migration of banded kokopu, ultimately resulting in reduced recruitment of juveniles in turbid rivers (Richardson et al. 2001). Other field studies have noted reduced densities of juvenile koaro (*Galaxias brevipinnis*), inanga (*Galaxias maculatus*), and banded kokopu in highly turbid waters following floods (McDowall & Eldon 1980), and within catchments containing high production pasture compared with forested streams (Minns 1990, Hanchett 1990, Rowe et al. 1992, 1999, 2000, Schicker & Boubeé 1990). Research on 38 East Cape streams by Richardson & Jowett (2002) revealed fish abundance and diversity reduced from 9 to 2 species as sediment loads increased. Streams with higher suspended sediments were shallower with swifter stream habitat, finer substrate, and less fish cover (Richardson & Jowett 2002).

Increased turbidity is now a characteristic of many lowland reaches of New Zealand rivers, with suspended sediment concentrations primarily related to flow rate. This can be temporarily increased (over months to years) by changes in land use such as conversion of forest to pasture, or by landslides following rainstorms (Hicks & Griffiths 1992). Logging of forested catchments may result in changes to stream light levels, water temperatures, flow patterns, stream bank stability, and bed characteristics, and also increase the size and frequency of floods (Morgan & Graynoth 1978). With New Zealand having one of the highest conversion rates of forestry to pasture in the world (over 60%), impacts on native fish species may be profound (McDowall 1978). Reports of prodigious whitebait catches declining drastically over the past 100 years have been documented by Phillips (1924a cited in McDowall, 1978) with reported “cartloads” of whitebait coming from the Hutt River in the 1880’s. Catches such as this were common from all over New Zealand.

5.4 Current New Zealand focussed work – snapper (*Pagrus auratus*)

We have virtually no information at all on the direct effects of stressors such as sedimentation on fish in the New Zealand marine context. However, recent work by M. Lowe (not yet published) on juvenile snapper in estuarine nurseries does provide evidence of such effects being important. We include some provisional indications from that work here to demonstrate that such effects may be significant. Several lines of evidence were collected.

Juvenile snapper less than one year of age (50–100 mm fork length) were collected using opera house traps from seven northern North Island estuaries covering a perceived spectrum of environmental degradation (sedimentation, associated water turbidity). Concurrent measures were taken of suspended sediments (by weight), secchi distance, temperature, and salinity. A relative condition index (CI), as per Francis (1997), was calculated for each fish as: Relative Condition Index = Carcass weight / Expected carcass weight. A significant negative relationship was found with increasing suspended sediment load, from Rangaunu Harbour (Northland) as the most ‘pristine’ harbour having the highest average condition indices, through to the Waitemata Harbour as the least ‘pristine’ having the lowest. In addition, significantly higher levels of gill deformation which included epithelial hyperplasia (cell proliferation), shortening and fusion of lamellae, along with higher parasite loads were recorded with increasing sediments. These data suggested a negative mechanism is operating on juvenile snapper fitness (defined as a condition factor), related to suspended sediment loads.

In higher water clarity estuaries (e.g. Rangaunu, Mahurangi, Whangateau, Tamaki) pelagic prey dominated the diet, especially calanoid copepods such as *Paracalanus indicus* and the cladoceran *Penilia avirostris*, while in the more turbid estuaries (Manukau, Kaipara, Waitemata) diet was dominated by benthic prey. These findings suggest that increasing turbidity levels cause a change in feeding strategy from active (probably visual) selection of pelagic prey (zooplankton), to larger, slower moving benthic prey. Such changes may reduce the overall food supply available to juvenile snapper, and perhaps relative nutritional values, by reducing either their ability to visually pick zooplankton, and/or a reduction in the actual zooplankton assemblages available due to changing environmental conditions.

Feeding trials on juvenile snapper (50–90 mm fork length) were conducted to examine the effect of turbidity on feeding rates of live prey in laboratory tanks, with turbidity being manipulated through the addition of suspended silt/clay (< 64 microns; increasing turbidity levels of ≤10, 20, 40, 80, and 160 Nepholometric Turbidity Units (NTU)). Sixty mysid shrimps (an important dietary item for snapper) were added to each tank. After thirty minutes, the fish were removed, and the number of surviving mysid's counted. A clear decrease in foraging success was evident with increasing suspended sediment levels, ranging from 77% in the controls (ambient sea-water of ≤10 NTUs) through to *c* 8% success in the tanks with NTUs of 160 (equivalent to storm conditions). Longer term tank experiments conducted over one month utilizing the same experimental protocol, found sub-lethal effects including increased coughing and gulping at the surface, paler colouration, higher respiration rates and decreased activity. Higher weight losses and overall mortalities, along with significant increases in gill deformation (epithelial hyperplasia and fusion of the lamellae) were recorded for the higher turbidity levels.

These results collectively suggest that increased suspended levels in the New Zealand marine environment can, and do, have negative effects on individual fish and their fitness, which may ultimately translate into reductions in subsequent productivity at the level of populations and fish stocks.

5.5 Impacts on habitat formers

Indirect impacts on harvested species include the loss of habitat-forming species (Figure 18) that provide important functions, such as settlement habitats that provide for the transition from larval to benthic phases, juvenile nursery habitats, feeding grounds, and spawning functions. As the importance of specific habitats to different life stages of harvested species is generally poorly known in New Zealand, it is also difficult in turn to quantify how land-based impacts on these habitats affect the harvested species that rely on them. However, we do have some understanding of some of the links. We note that the relative importance of a habitat depends not just on what it is, but also the wider habitat landscape within which it is embedded. Following are some selected examples of such habitat formers, their role, and likely threats.

Figure 18: Some of the kinds of habitat formers found in the New Zealand coastal zone. From left to right; sponge, horse mussels with soft corals, and sponges and kelp. (Source: NIWA.).

5.5.1 Seagrass (*Zostera mulleri*)

From sampling of numerous estuaries, we know that subtidal seagrass provides an important nursery function for juvenile fish, including snapper (Figure 19), trevally, parore (*Girella tricuspidata*), garfish/piper (*Hyporhamphus ihi*), and spotties (e.g. Morrison & Francis 2003 Francis et al. 2005, Schwarz *et al.* 2006b).

Seagrass meadows have been affected impacted by human activities across New Zealand and shown large declines in spatial extent (Inglis 2003), as in many other regions of the world (Duarte 2002, Pihl et al. 2006, Lotze et al. 2006). Subtidal elements, by far the most important component for juvenile fish, have been particularly affected. Possible mechanisms include increased direct sedimentation, reductions in light levels in the water column, and over-growth by epiphytic algae benefiting from higher nutrient levels from land-off preventing adequate light reaching seagrass blades (Turner & Schwarz 2006). Large scale losses have been documented from the Whangarei, Waitemata, Manukau, Tauranga, and Avon-Heathcote estuaries (Inglis 2003). Whangarei estuary (east coast, Northland) completely lost 12–14 km^2 of seagrass, much of it subtidal, in the late 1960s following the dumping of 5 million tonnes of sediment 'fines' into the estuary from port expansion and a cement works (Morrison 2003).

Figure 19: A subsample of a fine-mesh beach seine fish sample from Rangaunu Harbour (east coast, Northland), which included almost 2 000 juvenile snapper, along with a high abundance of other species. Average juvenile snapper catches across multiple sample tows were 160 and 20 juvenile snapper per 100 m^2 (Morrison et al., unpublished data) for two different locations in the harbour. (Source: Paul Buisson, DOC.)

Park (1999) used aerial photography to quantify seagrass loss in Tauranga Harbour between 1959 and 1996 (Figure 20), and also found an overall 34% decline in seagrass cover across the whole harbour over this period.

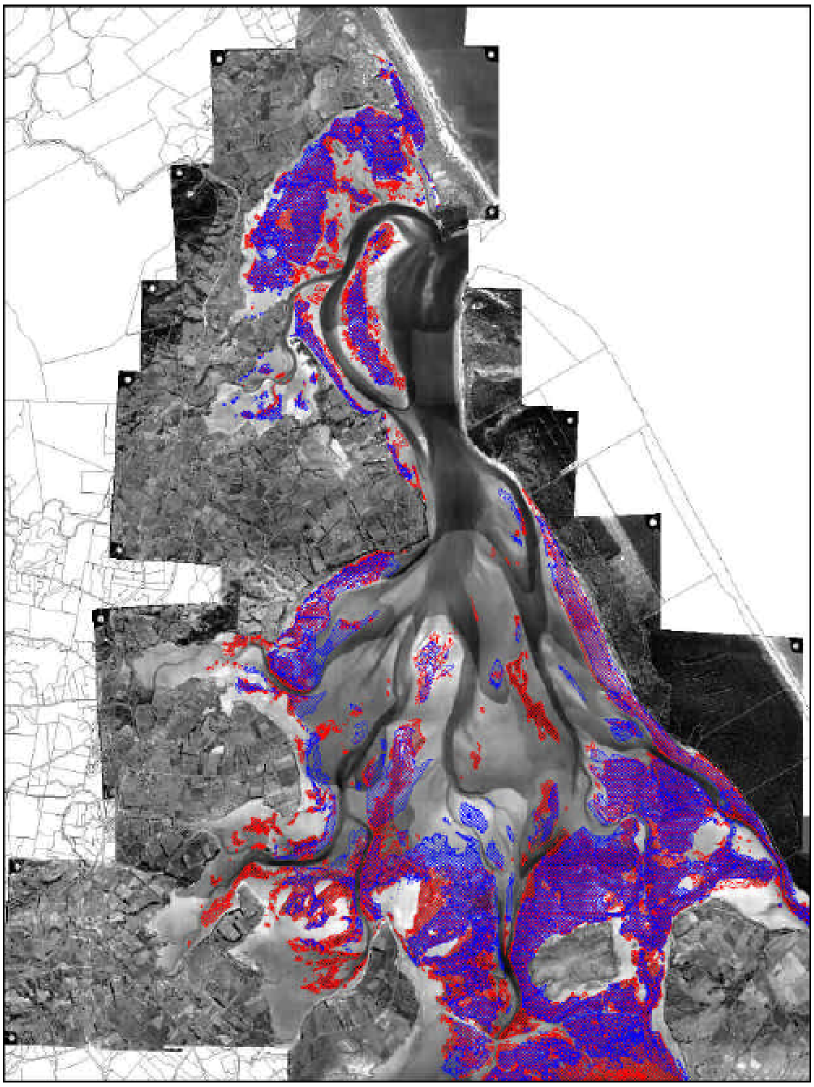


Figure 20: Northern Tauranga Harbour showing the presence of seagrass in 1959 (red) and 1996 (blue). (Source: Figure 3.1 from Park (1999), Environment Bay of Plenty.)

Seagrass beds in the shallow subtidal and sub-estuary areas with large catchments suffered the most, with 90% of all subtidal seagrass being lost. The areas near the harbour entrance with little land run-off or influence from other catchments showed the smallest decline in seagrass abundance. Rates of loss in the sub-estuaries were well correlated with suspended sediment loadings into these areas on the basis of relative area ($r = –0.869$, $p = 0.005$). There was also a reasonably strong, but not statistically significant, negative correlation with the mud content of sediments, and nutrient loads of P and N coming from the catchments (r = –0.773, –0.755, and –0.740 respectively). Earlier analysis of seagrass distribution in Tauranga Harbour found that seagrass was generally absent once the mud content of surface sediments reached 13% (Park 1994, in Park 1999). Overall, the evidence collected by Park (1999) strongly pointed to sediment and nutrient run-off as the main factors involved in seagrass loss. He noted that substantial losses might have already occurred before 1959. However, there was some recent evidence for some possible recovery, as seen in places such the Tuapiro sub-estuary.

5.5.2 Horse mussels (*Atrina novaezealandiae*)

Horse mussels are highly sensitive to increased sedimentation loads (Ellis et al. 2002, Hewitt & Pilditch 2004, Lohrer et al. 2006b), with documented population declines from declining environmental quality in the Mahurangi Harbour (Figure 21) on the east coast of north New Zealand (Cummings et al. 2005). The ecology of this harbour is known to be changing as a result of increased sediment loading (Gibbs 2004). Recent 'forensic tracking' of sediment into the harbour from three land use types (rural pasture, native forest, exotic pine forest) onto the open mud-flats into the upper harbour, and into mangrove forests, suggests through a mixing model that 50–54% may be derived from exotic forest, 32–44% from pasture, and 5–14% from native forest (the latter two estimates being less certain). Large tracts of exotic forest were planted in this catchment in the 1970s, and these are now being harvested and replanted (Gibbs 2004). Most sediment comes from a small number of storm events each year.

The Mahurangi Harbour is a known juvenile snapper nursery, with an estimated juvenile snapper (under 100 mm) population of 105 000 ± 17 000 in 2004 (Morrison & Carbines 2006) (NB: this species shows strong year-class strength variations). These juvenile snapper are associated with seafloor structure, dominated by horse mussel beds and associated epifauna. Current PhD work by N. Usmar (Leigh Marine Laboratory, University of Auckland) is examining snapper habitat ontogeny in this harbour, including the importance of horse mussels.

Figure 21: Mahurangi Harbour, a sheltered estuary in the northern Hauraki Gulf. (Source: Natalie Usmar, Leigh Marine Laboratory, University of Auckland.)

Horse mussels occur around much of the New Zealand coast, and have been recorded to depths of at least 100 m off the east coast of the South Island (Vooren 1975). It is suspected that they may also play an important nursery habitat role for other harvested fish species, with associations having been found with juvenile tarahiki off the South Island, as part of a wider biogenic habitat mixture of sponges and bryozoans (Vooren 1975). The habitat usage of very small blue cod is also poorly known, and it is quite possible that this species may recruit to biogenic habitats (including horse mussels) adjacent to rocky reefs (Figure 22), and then move with increasing size onto the reefs. Juvenile forms (about 3–5 cm) have been seen off Goat Island Bay, Leigh Marine Reserve, in association with horse mussels and sponge (M. Morrison, pers. observ.), and in association with kelp at Hahei Marine Reserve, Whitianga

(N. Usmar, pers. observ.). If this proves to be an important habitat for such species, the potential for land-based impacts to affect such relationships will depend on where such nursery habitats are found, relative to the potential for such effects to reach them.

Figure 22: Juvenile blue cod, estimated to be *c.* 7–8 cm long, on horse mussel bed on sand, probably in about 25–30 m water depth. Goat Island Bay. (Source: Grant-Mackie (1987).)

5.5.3 Sponges

Sponges are an important component of many coastal environments, especially on rocky reefs, and on coarser soft sediments in higher current areas. Their relative dominance of reef environments increases with depth, as kelps reach the limit of their light tolerance. As filter-feeders, they are sensitive to increased suspended sediment loads. Work by Lohrer et al. (2006b) found that the sponge *Aaptos* spp (a circular species found on rocky reefs), showed declines in condition relative to controls after experimental exposure to fine sediment deposits for three weeks, with water filtering rates also declining by about 40%. Sponge gardens have been reported to possibly provide nursery functions for juvenile snapper on northeastern New Zealand reef environments (Battershill 1987), and current work on soft sediment systems in the inner Hauraki Gulf is showing strong relationships between the abundance of snapper (especially juveniles) and the presence of biogenic seafloor structure, including horse mussels, and sponge species such as the yellow finger sponge, *Callyspongia ramosa* (Morrison et al. 2008).

5.5.4 Mangroves (*Avicennia marina*)

Mangroves are one of the few habitat-forming species to show substantial increases in some areas, in response to land-based effects such as increased sedimentation and nutrient levels. Their expansion is causing strong societal debate as to their ecological values versus the loss of human amenities such as sea views, swimming, boating, and fishing. A comprehensive technical review of what is known about New Zealand mangroves and their ecological role has been completed by Morrisey et al. (2007). Their importance as juvenile fish nurseries has formed part of this debate. Small fish were sampled across eight estuaries to address this information gap. Eight estuaries, perceived to encompass an environmental cline of degradation, from clear waters with seagrass meadows abutting mangrove forests (Rangaunu Harbour), through to highly turbid harbours with high suspended sediment loads (e.g., Manukau and Kaipara), were sampled using fine mesh fyke nets. Using the definitions of a fish nursery ground given by Beck et al. (2001), and Dahlgren et al. (2006), only three fish species were considered to use mangrove as nurseries. These were short-finned eels (*Anguilla australis*) on both the east and west coasts, grey mullet (*Mugil cephalis*) on the west coast, and parore on the east coast. None of these species were completely reliant on mangroves as nursery habitats, with each also using alternative habitat types.

5.5.5 Kelp/seaweed forests and assemblages

Kelp/seaweed forests are a fundamental habitat element on shallow water rocky reefs around New Zealand (Choat & Schiel 1982, Schiel & Foster 1986, Shears & Babcock 2004, 2007). Like seagrass, they are vulnerable to changes in light regimes, as well as siltation of surfaces for spores to settle on. For instance, lab-based experiments by Schiel et al. (2006) showed that a light dusting of sediment reduced zygote attachment rates of Neptune's necklace (*Hormosira banksii*) by 34%, and of bull kelp (*Durvillaea antarctica*) by 71%, while a complete cover of sediment completely prevented attachment. Such species provide important habitat for a range of invertebrate and finfish species, including in some cases as juvenile nurseries, though our knowledge remains geographically limited. For instance, Neptune's necklace is a known important nursery habitat for juvenile parore in north-eastern New Zealand in some estuaries, such as in the Whangateau Harbour, Northland (just south of Leigh) (Figure 23). Juveniles settle directly into these and other estuarine habitats (subtidal seagrass beds, mangrove forests) from the plankton, where they spend their first 3–4 months, followed by dispersal to other nursery habitats such as estuarine *Carpophyllum* kelp forests (Morrison 1990).

Figure 23: Juvenile parore (*Girella tricuspidata*) in association with Neptune's necklace beds at Horseshoe Reef, Whangateau Harbour Northland (March 2008). (Source: Natalie Usmar, Leigh Marine Laboratory, University of Auckland.)

Juvenile snapper are also associated with kelp forests on the edge of shallow reefs (Ross et al. 2007), and with algal turfs (Kingett & Choat 1981) though their relative contribution to adult populations is not yet known.

Schwarz et al. (2006a) examined kelp and invertebrate patterns along a gradient of increasing water turbidity from Hahei towards the Whitianga Harbour mouth in Coromandel, New Zealand. At the site closest to the harbour mouth, the brown kelp *Ecklonia radiata* exhibited the greatest degree of acclimation to low light levels for a given water depth. There was also evidence for a lower plant density at this site, though overall plant biomass was not significantly different from equivalent depths at the other three sites, suggesting plants were compensating in some way. It was suggested that the reduced photosynthetic potential observed might reflect reduced production in the general primary producer assemblage, including epiphytes associated with *E. radiata*. At the same innermost site, the lowest density, biomass, and productivity of epifauna (e.g., crustaceans, gastropods and polychaetes) on *E. radiata* was also found. As these epifauna are thought to be responsible for about 80% of the flow of energy and materials through rocky reef animal communities, it was suggested that the effects of increased suspended sediment concentrations may have serious consequences on energy flows on New Zealand's rocky reefs, where human-induced increases in suspended sediment concentrations occur (Schwarz et al. 2006a).

5.5.6 Other potentially important habitats

There are a number of other potentially important habitats, about which we know very little, both of what harvested species may use them at some stage during their life cycle, and of the threats of land-based activities to those habitats. One of these is maerl (also known as rhodoliths) (Figure 24), which are calcareous red algae that often form nodules around small objects such as shells or stones, and which in aggregate form loose beds. Such habitat is not uncommon in New Zealand waters, but we know very little about them. Limited sampling of beds off Kapiti Island (Wellington coast) has found a high diversity of associated invertebrate species, along with the presence of 0+ blue cod (W. Nelson, NIWA, Wellington, pers. comm.).

Figure 24: Example of a maerl bed, 5 metres water depth, at Elizabeth Reef, Kawau Bay, northern Hauraki Gulf. Goatfish (red mullet) are in the upper image; 10 cm below this bed is a dense dog cockle bed.

Large beds have been documented in Northland off the Cavalli Islands (Grace & Hayward 1980), and Urapukapuka Island, Bay of Islands (Hayward et al. 1981). These species seem to be associated with areas of higher current flows, especially around islands and headlands with strong tidal currents and relatively clear water conditions. As these are photosynthetic algae, they are very vulnerable to loss from habitat degradation through sedimentation and reductions in light levels, as well as being susceptible to direct damage from physical interactions with fishing gear and anchors. We have minimal knowledge of where these species occur, how physical factors drive their distribution and abundance, and their role as juvenile invertebrate and finfish nurseries.

However, a significant amount of research has been done on maerl in European waters (Airoldi & Beck 2007). Some beds have been dated to older than 5 500 yrs (Grall & Hall-Spencer 2003). They occur in exposed and sheltered environments down to 100 m water depth, and are most typically found at 20–30 m (OSPAR Commission 2005). They are considered highly sensitive to overexploitation and other human activities that result in physical disturbance or deterioration in water quality (Barbera et al. 2003), especially smothering by fine sediments (Wilson et al. 2004). With extremely slow growth rates (about 1 mm yr^{-1}) substantial deposits take centuaries to millennia to accumulate (Hall-Spencer et al. 2003). Negative effects of increased eutrophication and turbidity in coastal waters, both from silt loads and nutrient run-off from agricultural land and aquaculture, have been well documented in Galicia and in the Bay of Brest (Barbera et al. 2003).

Tubeworms may also occur in such densities as to create mounds and other structures. Smith et al. (2005) examined 114 subtidal *Galeolaria hystrix* (Polychaeta: Serpulidae) patch reefs (Figure 25) in Big Glory Bay, Paterson Inlet, Stewart Island, ranging from 1 to 5 m in diameter, and up to 1.5 m high (surveyed in 1995 and 2000). The remnants of one dead reef were 100 m in diameter. Up to 65% of the tubes were occupied by worms, with 64% of reefs in a whole state, and the remaining 36% broken or dead. Radiometric dating of a basal reef portion returned a date of less than 50 years. Fish seen in association with these reefs included blue cod, spotties, pigfish (*Congiopodus leucopaecilus*), red cod (*Pseudophycis bachus*), butterfly perch (*Caesiperca lepidoptera*), and triplefins (family Tripterygiidae), while skate (*Raja nasuta*) and shark egg cases were commonly found attached to the reefs.

Figure 25: A tubeworm (*Galeolaria hystrix*) reef in Big Glory Bay, Paterson Inlet, showing large numbers of associated spotties. Spotties are one of the most easily observed components of fish assemblages that associate with seafloor structural complexity. (Source: A. Smith, University of Otago.)

Overall, the diversity and abundance of secondary fauna (invertebrates, algae, and fish) associated with the reefs was observed to be greatest in those reefs with the highest occupancy of worm tubes (Figure 26). It was noted that while inner Glory Bay reefs were largely in '*excellent*' condition, they had a '*great deal of sediment on them*', and that sponges and ascidians were generally absent. Divers described the general environment as "*quite silty*", with the surrounding sediments being terrigeneous mud. As noted by Smith et al. (2005), tubeworms are suspension feeders, and excessive sedimentation may lower fecundity, cause damage, or even kill them (Kupriyanova et al. (2001), but see Frank & ten Hove 1992). Deeper areas of *Lenormandia* algal meadows, a macro-algae species that lives on soft sediments, were also mentioned.

Figure 26: A tubeworm (*Galeolaria hystrix*) reef in Big Glory Bay, Paterson Inlet, showing the diversity of associated invertebrate species. (Source: A. Smith, University of Otago.)

5.5.7 Habitat landscapes

Habitats do not operate in isolation from each other, but rather as a habitat landscape. The relative abundance and spatial configurations of different habitats, as well as habitat quality, are important factors in driving what associated species will be produced from that habitat landscape, including harvested species. One of the effects of land-based impacts may be to alter the configuration of these landscapes. For example, in estuarine environments, mangroves and muddy substrates with low water clarities may increase, while other habitats such as seagrass meadows, and sandy substrates with high water clarities, may decline. In turn, this will favour some species and disadvantage others, depending on their habitat requirements. Saintilan (2004) examined the relationships between the weight of commercial fish landings and physical characteristics at the estuary scale, using data from 55 estuaries along the coast of New South Wales, with a focus on seagrass and mangroves. While a correlative study, he found that the role of mangroves in supporting commercial fisheries was modest in this temperate region, and that "*as estuaries infill and the area of seagrass and mud basin declines, so too does the catch of species dependent upon these habitats*", and that "*the results strongly suggest that seagrass is a critically important habitat for a range of commercially important species, and that declines in seagrass area resulting from natural or anthropogenic disturbance should lead directly to decreases in stocks of these species*" (Saintilan 2004). Such landscape effects will also be operating in the New Zealand environment, and work at the habitat landscape level would be valuable.

6. PLACES WHERE IMPACTS ON IMPORTANT FISHERIES RELATED HABITATS HAVE BEEN INVESTIGATED

6.1 Whangapoua Estuary, Coromandel Peninsula – seagrass nurseries for snapper, trevally and other fish species

Whangapoua Estuary is located 20 kilometres north of Whitianga on the eastern side of Coromandel Peninsula. It has an area of 10.8 km^2, with an associated catchment of 107 km^2. Current land use in the catchment is 54% exotic forest, 20% native forest and scrub, and 17% pasture (Jones 2008). The native forest and scrub is present only on the very steep ridges above the pine forests, while the pasture areas are found on the valley floor. The estuary has a large tidal prism relative to the catchment area, with 80% of the estuary being intertidal, and supporting extensive seagrass beds, salt-marsh and mangroves. It is classified in the Waikato Regional Coastal Plan as an Area of Significant Conservation Value (ASCV) (Jones 2008). Fish sampling of the seagrass meadows and adjacent bare sediment flats has demonstrated that the presence of seagrass, even just small 'scrubby' subtidal patches, significantly increases the abundances of juvenile snapper, trevally, parore, pipefish, and triplefins, relative to non-vegetated areas (Morrison & Francis 2003). Aerial photographs from 1945 indicate that historically the extent of seagrass meadows in this estuary was much more extensive, including covering much of the subtidal channel areas, which extend down to 5 m in depth.

Overall, mangrove cover has more than doubled, and seagrass cover more than halved, over the last 50 years (Jones 2008). Subtidal seagrass elements have experienced much larger losses proportionally than intertidal beds. Since 1993 there have been gradual declines in the abundance of invertebrate species known to be sensitive to sedimentation (Hewitt 2001, in Jones 2008). A major storm in March 1995 demonstrated the impact that such events can have on seagrass meadows and their associated fauna (Morrisey et al. 1995). At all sites being monitored in the harbour, there was sediment deposition, with a layer of orange mud up to 10 cm thick in some areas (Figure 27). Silt appeared to have been distributed throughout the estuary within six days of the storm, with deepest deposits in the upper parts of the Owera Arm. There was little associated woody debris. Large areas of seagrass were partly or completely covered by silt, with sediment below the surface being anoxic (depleted in oxygen). Large numbers of dead cockles were present in and around the seagrass, along with a few dead crabs, mantis shrimps and snapping shrimps (Morrisey et al. 1995). Aerial photographs from 1993 to 1997 showed a complete loss of seagrass from some areas related to this storm, though it was also noted that this was not unprecedented given the photographic record from 1945 to 1995 (Morrisey et al. 1999). Seagrass leaf density, a measure of seagrass health, decreased through time within those beds not already eliminated by the sediment. It was suggested that this might have occurred via a mechanism where water clarity decreases were caused by the progressive removal of fine sediments from sand-flats and seagrass beds by wave action (Morrisey et al. 1999). Overall, the critical factor appeared to be sediment yield from forested areas (both absolute, and relative to alternative end-uses) in response to such occasional large events, rather than chronic yields in response to small, more frequent events (Morrisey et al. 1999). A review of the monitoring in 2006 found a statistically significant relationship between the extent of forest harvesting (clear-felling) by sub-catchment, and the decline in benthic assemblages sensitive to increased sedimentation in associated arms of the estuary (Halliday et al. 2006, in Jones 2008).

Figure 27: Aftermath of the March 1995 storm in Whangapoua Harbour; a) cockle bed smothered by sediment, b) and c) seagrass beds with heavy sedimentation, d) seagrass bed covered by mud-laden water. (Source: Don Morrisey, NIWA.)

The consequences of such events are both direct (mortalities of harvested species such as cockles, and indirect (reduction in the carrying capacity of fish nursery habitats for significant periods of time).

6.2 Separation Point bryozoan mounds – nursery grounds for snapper, tarakihi, and John dory

Between Golden and Tasman Bays, off the Abel Tasman National Park, are the Separation Point bryozoan beds, covering some 55 km^2 of seafloor in 2003 (Figure 28) (Grange et al. 2003). These are protected within a larger 156 km^2 zone, where all power-fishing methods have been banned since 1980 (Mace 1981). Additional bryozoan habitat areas occur off D'Urville Island, and parts of the outer Marlborough Sounds. Trawl sampling in the 1970s identified these habitats to be important juvenile fish habitats for snapper, tarakihi, and John dory (Vooren 1975). Prior to this, these beds were known for catches of juvenile snapper and tarakihi by local fishers, who gave them the nickname “the Coral Grounds”. In addition to their fish nursery value, they also increase seafloor habitat complexity and biodiversity values. Before 1956, such areas were avoided by fishers as the cotton trawl nets available were easily damaged through contact with bryozoans. However, following the design of special nets which floated just above the seafloor, fishers targeted the beds off Torrent Bay (Grange et al. 2003). The Torrent Bay beds were dominated by the more brittle bryozoan species *Hippomenella vellicata* (‘paper coral’) (Figure 29). As these beds were impacted and declined, so did the proportions of juvenile fish (snapper and tarahiki) in the catch (Saxton 1980a). At Separation Point, the more robust species *Celleporaria agglutinans* dominated, which protected this area from fishing until 1972–74, when the introduction of pair trawling (trawl net towed between two vessels, heavy otter boards not required) enabled fishers to fly their nets over the area (Grange et al. 2003).

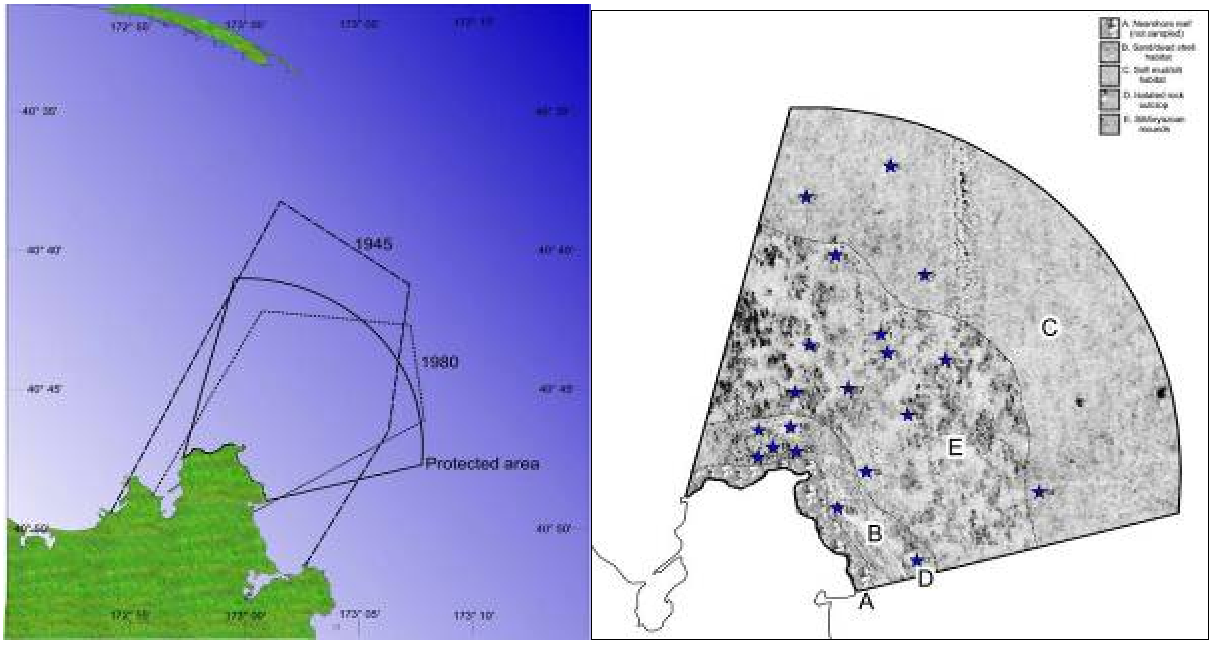


Figure 28: Left: Location of Separation Point bryozoan bed protected area, and extent of the beds themselves as in 1945 and 1980, based on fisher observations. Right: digital side-scan imagery mosaic, showing the present day bryozoan bed extent (habitat ‘E’). (Source: Ken Grange, NIWA.)

Significant damage to the bryozoans commenced, and by 1979 there was concern among fishers that these beds would be destroyed (along with their important fish nursery functions), as they observed reductions in the numbers of juvenile snapper and tarakihi (being caught (Saxton 1980a). In 1980, a 156 km^2 area of seafloor was closed to power-fishing methods, including an estimated 118 km^2 of bryozoan beds (reduced from an estimated 213 km^2 in 1945) (Saxton 1980a, 1980b). In 1982, limited diving observations were made in the area, and *C. agglutinans* was found to cover up to 50% of the seafloor in some places, with colonies to 50 cm high. The area was characterised as being very turbid, with very low levels of light penetration and considerable tidal currents (Bradstock & Gordon 1983).

Some two decades later, the full extent of the protected area was mapped using side-scan sonar, and selected sites ground-truthed using a remote operated vehicle (ROV) (Grange et al. 2003). The contemporary mound cover was estimated at 55 km^2, covering 38% of the protected area. There was also the suggestion of bryozoan mounds occurring outside the protected areas boundaries. Each ROV station covered some 10–25 m^2 of seafloor. These mounds included many species of bryozoans, as well as brachiopods (*Liothyrella neozelanica*), sponges (e.g., *Callyspongia*), hydroids, and horse mussels (Figure 35). The ROV sighted barracouta (*Thyrsites atun*), tarahiki, and leatherjackets (*Parika scaber*). No evidence of trawl or dredge marks was evident on the side-scan records, which are capable of detecting such fishing marks.

However, many colonies appeared to be growing only from the distal tips, and were covered by a film of silt, suggesting the community may be stressed by sedimentation (Grange et al. 2003). Samples of C. *agglutinans* taken from a dredge tow confirmed this observation of growth occurring only on the distal tips, with the lower portions being devoid of living tissue (identifiable by its pink coloration). Batson & Probert (2000) listed the environmental parameters at seven New Zealand locations where bryozoan dominated sediments have been documented. Bryozoan mounds are rare in silty environments. All of the locations discussed by Batson & Probert (2000) were situated on biogenic/carbonate sediments, with associated strong tidal currents and high energy environments, suggesting low sedimentation regimes (similar characteristics are associated with the recently discovered biogenic assemblages of Spirits Bay (Cryer et al. 1999)). Although tidal currents are reasonably strong around Separation Point (about 30 cm s^{-1}), the seafloor is dominated by soft muds and silts, suggesting high sedimentation rates. It is possible that originally bryozoans may have established on coarser sediments, before heavy sedimentation over these sediments following human-driven forest clearance. It was also speculated on that if considerable damage had occurred to the main-habitat forming mounds before protection, they would not have been able to recolonise the soft muds present in the area today (Grange et al. 2003). The Torrent Bay area appears not to have recovered, and this was attributed to the frame-building mounds being unable to recover on the soft mud sediments once they were broken up (Grange et al. 2003).

A hydrographic model has been produced for Tasman and Golden Bays which is able to match the gross features of the major tidal flows in this region (Tuckey et al. 2006). By running the model with a range of different river flows and wind directions and velocity, it was shown that the Motueka River influence can cover considerable proportions of the western side of Tasman Bay, and during flood conditions can extend around Separation Point and into Golden Bay. A sediment transport model run in tandem with the hydrographic model, and representing the patterns of fine sediment entering the bays from the four principal rivers, produced results which were consistent with existing bathymetric and seabed substrate characteristics. Model runs with 20 knot winds in either a northeasterly or southeasterly direction, combined with a river discharge rate of 1000 m^3 s^{-1} (within the range of empirical measures from the system), showed transport of fine sediments to Separation Point and Golden Bay. Most of the suspended sediments inputs to the bays occur during storm events.

The loss of areas of these biogenic structures has almost certainly reduced overall finfish productivities in the surrounding regional ecosystem (and perhaps beyond) for some fished species (e.g., snapper, tarahiki, and leatherjacket). There are also broader issues of reduced habitat complexity and associated biodiversity. With the current assemblages being under stress from sedimentation, and limited recovery potentials if further damaged, these issues deserve more attention.

Figure 29: Bryozoan mounds at Separation Point, including a) hydroids and sponges, b) bryozoan (*Hippomenella*) and encrusting fauna, c) bryozoan (*Celleporaria*) and hydroids, d) bryozoan colony (*Celleporaria*). (Source: Ken Grange, NIWA.)

7. CASE STUDIES ACROSS COASTAL ECOSYSTEMS

7.1 Habitat connectivity over large spatial scales

Many coastal adult fish populations use habitats and areas that are spatially discrete from those of their juvenile populations (Beck et al. 2001, Gillanders et al. 2003). Over the lifetime of any given individual fish, a sequential ‘chain’ of habitats may be used, including some that have the potential to act as “habitat bottlenecks” for juveniles, resulting in overall lower population sizes, and associated adult biomass (Werner & Gilliam 1984). Human impacts on these habitat bottlenecks (from land-based or marine-based impacts) may have very strong negative cascades into overall adult population sizes. In New Zealand’s coastal ecosystems, we have only a very rudimentary understanding of such dynamics, and such connectivity is not considered in management regimes beyond large scale stock units. There are a few exceptions, such as the protection of the Separation Point bryozoan mounds – although we do not yet know the true value of these habitats for juvenile fish i.e. the proportions and spatial extent of their contributions to surrounding adult populations.

For two finfish species, we do have some understanding of this connectivity: snapper on the west coast of the North Island and sand flounder around the Canterbury estuaries and adjacent coast.

7.2 West Coast North Island snapper

The west coast snapper fishery (SNA 8) is currently dominated by contributions from only a few year classes (3–7 year olds), with most of the larger and older fish having been removed over time. This implies that several sequential years of low juvenile recruitment could result in the fishery becoming uneconomic due to very low population densities. Before this, a large reserve of many age classes would have ‘buffered’ the stock for several decades. Recent research has demonstrated the central importance of the Kaipara Harbour to the wider west coast ecosystem (Morrison et al. unpublished. results). Sampling of juvenile snapper (Figure 30) in the seven main west coast estuaries (Whangape–Kahwia), and of snapper population/s along the coast, suggested a movement model in which the Kaipara estuary provides most of the juvenile snapper to the open coast, with a subsequent dispersal with time and age of fish away from the Kaipara estuary coastline. This model was strongly supported by otolith chemistry, which compared the chemical concentrations of 0+ snapper otoliths collected in the estuaries (2003), with the same cohort (age class) of fish re-sampled in 2007 from the coastal commercial fishery. Using the estuarine juvenile fish as a ‘signature library’, the large majority of the 140 adult fish sampled from the commercial fishery were assigned to the Kaipara Harbour as their natal nursery estuary. These adult fish came from five spatial zones covering the entire west coast, ranging from Ninety Mile Beach down to Mana Island, just north of Wellington.

Figure 30: A beam trawl catch from the Papakura Channel, Manukau Harbour. Along with juvenile snapper, the catch included juvenile red gurnard, adult triplefins (a small goby-like fish), and juvenile sand flounder (which dominated the sample).

Sampling inside the seven estuaries using a fine mesh beam trawl found juvenile snapper (30–100 mm length) were relatively abundant in the estuaries (230–11 000 per km^2, depending on harbour), and were strongly correlated with the presence of horse mussel beds (indexed by horse mussel bycatch) and terrestrial debris (Whangape estuary). Additional sampling of shallow subtidal seagrass by beach seines in the Kaipara Harbour also found high abundances (70 000 per km^2). Such fish were rare on the open coast (sampled by Ministry of Fisheries otter trawl surveys), and where present, occurred either adjacent to the estuary entrances or along parts of Ninety Mile Beach, at lengths of 10–15 cm (10–12 months old). These low coastal fish densities (Figure 31) probably represent early migrants from the estuaries, especially those spatially adjacent to the estuary entrances, although there may be a modest juvenile snapper nursery off Ninety Mile Beach.

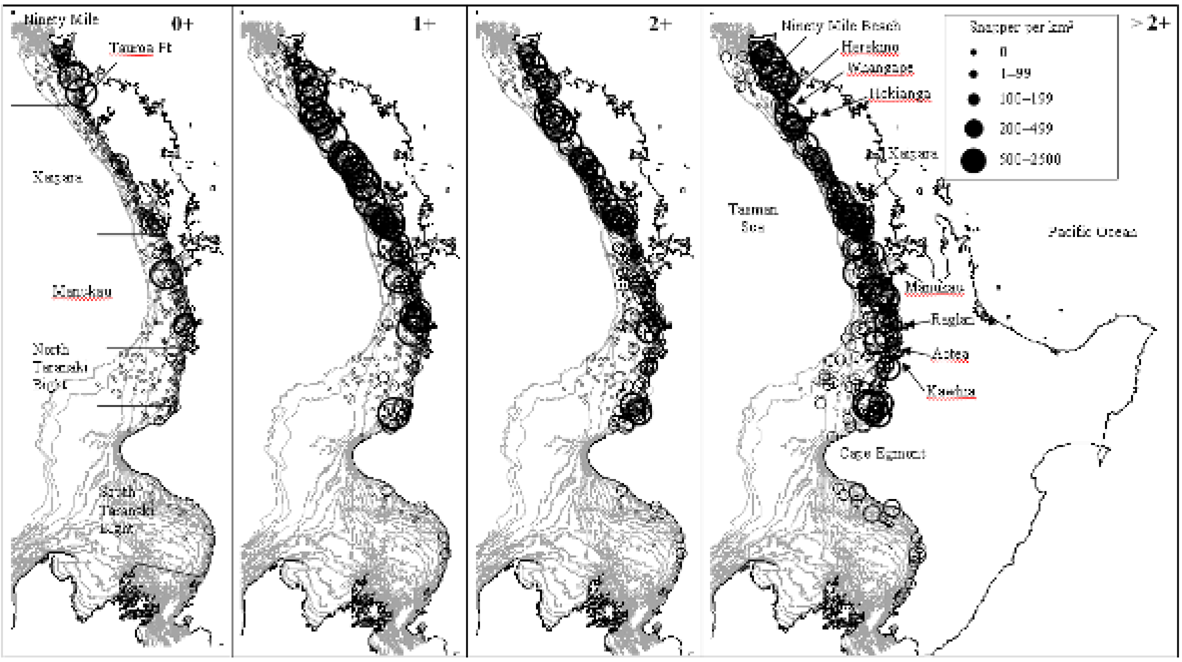


Figure 31: Snapper densities per km^2, by age class (0+, 1+, 2+, and >2+), along the west coast of the North Island, as sampled by Ministry of Fisheries trawl surveys. Positions of estuaries sampled are shown (except Herekino estuary). Also shown are the five coastal sampling zones from which adult snapper were collected for otolith analysis (the majority of which were assigned to the Kaipara Harbour as their natal nursery).

Most snapper juveniles sampled within the estuaries were associated with the presence of live horse mussels. These beds occurred in spatially discrete patches/strips on the edge of the main subtidal channels (Kaipara and Manukau estuaries), or over relatively extensive subtidal sand-flats in 2–4 m water depth (Hokianga estuary). Juvenile snapper were also strongly associated with subtidal seagrass meadows in the Kaipara estuary. Overall, seagrass extent in the Kaipara is thought to be about 15 km^2, most of it is inter-tidal, but with substantial subtidal elements. Subtidal seagrass meadows are not known to occur in any of the other estuaries sampled. Positive juvenile snapper associations with seagrass have previously been shown for northern New Zealand (Francis et al. 2005, Schwarz et al. 2006b). In Whangape estuary, the presence of terrestrial debris (logs and branches) was associated with higher snapper catches, a feature also seen in some east coast estuaries (e.g., Fitzroy Harbour, Great Barrier Island, M. Morrison, pers. obs.). In the Aotea estuary, an inadvertent tow over an unmarked small patch reef with a diverse sponge and ascidian assemblage produced most of the 0+ snapper from that estuary. Overall, it appears that the presence of structurally complex benthic habitats (irrespective of specific identity) in estuarine environments is consistently associated with higher densities of 0+ snapper, relative to surrounding 'bare sediment' habitats.

Figure 32: Juvenile snapper and parore caught by beach seine in Rangaunu Harbour, east Northland. The two size cohorts in the sample represent multiple successful spawning events for that year. (Source: Crispin Middleton, NIWA.)

As discussed in Section 5.5, many of the complex three-dimensional habitats in which juvenile snapper were found in association with such as horse mussels, seagrass, and sponges (Figure 32) are biogenic (living), and hence vulnerable to the effects of environmental degradation arising from human activities, both in the estuaries themselves, and the surrounding land catchments. For instance, for the west coast ecosystem, seagrass meadows more than a mile (1.6 km) wide were once present in the Manukau estuary, as described by Morton & Miller (1973). These no longer exist, and this estuary currently contributes less then 2% of snapper to the coastal population based on otolith chemistry results. It probably historically once played a much more important role. Essentially, the Kaipara Harbour now appears to sustain most of the adult coastal populations. Its habitats are known to be under increasing pressure from a range of stressors (Haggitt et al. 2008), especially those derived from land-based human activities, such as increased sedimentation that adversely affect habitat elements such as horse mussels and seagrass beds. Any negative impacts on the Kaipara estuary's production of juvenile fish will cascade through into the much larger coastal ecosystem, ultimately affecting the abundance of fish several hundreds of kilometres away, and coastal snapper fisheries along the entire coast will be severely impacted. Such ecosystem linkages need to be explicitly incorporated into our management of estuarine and coastal ecosystems, including fisheries (Morrison et al. unpubl. results).

7.3 Sand flounder in the Christchurch region – Canterbury Bight and Pegasus Bay

Between 1964 and 1966, more than 15 000 sand flounder (15 cm to over 30 cm; 87% less than 25 cm) were externally tagged in three Christchurch region estuaries (Avon-Heathcote, Lyttelton, and Akaroa) (Figure 33) (Colman 1978). Recaptures were recorded from recreational and commercial fishers, as well as some by researchers: 3 307 tagged fish were returned with sufficient associated location data to be used to assess movement.

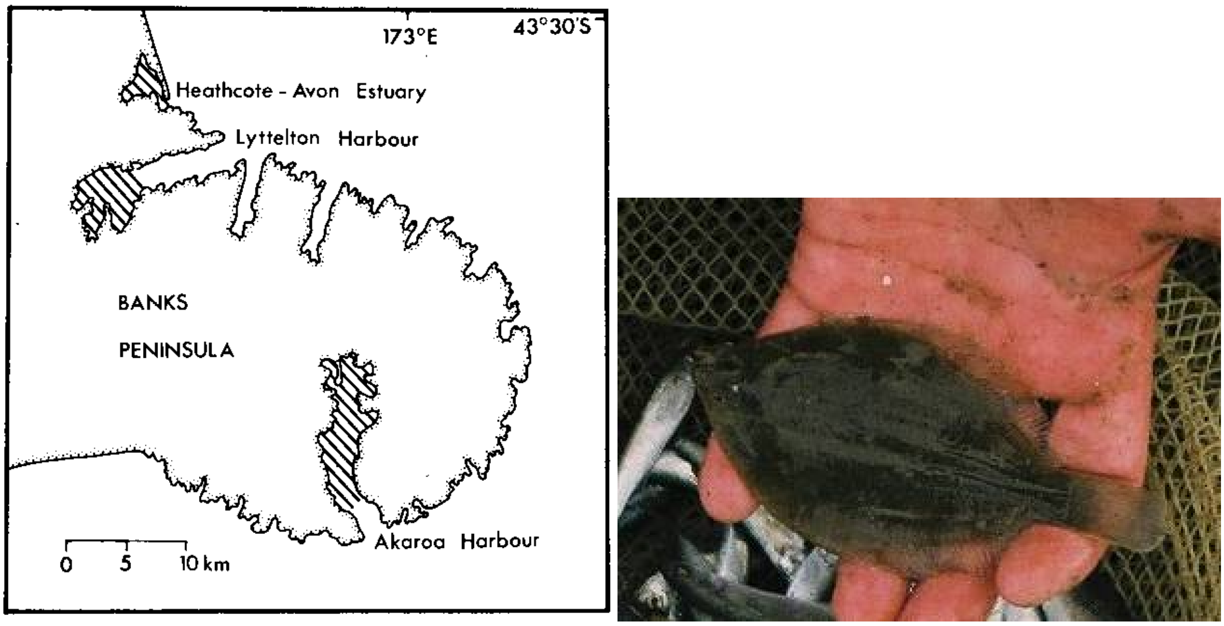
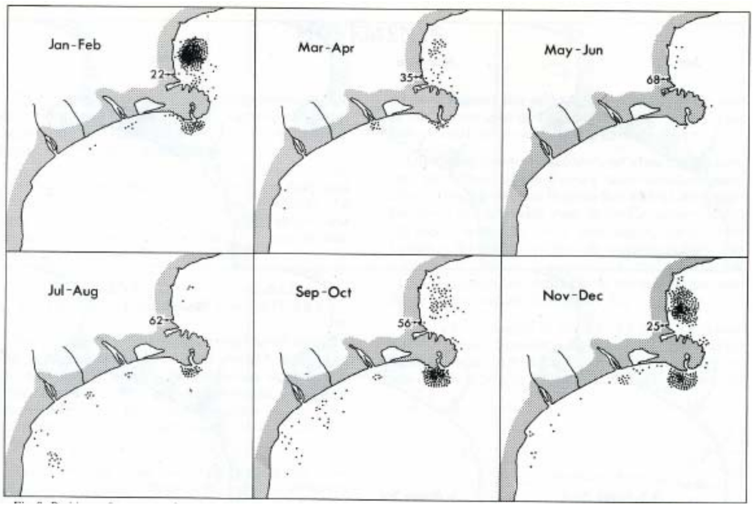
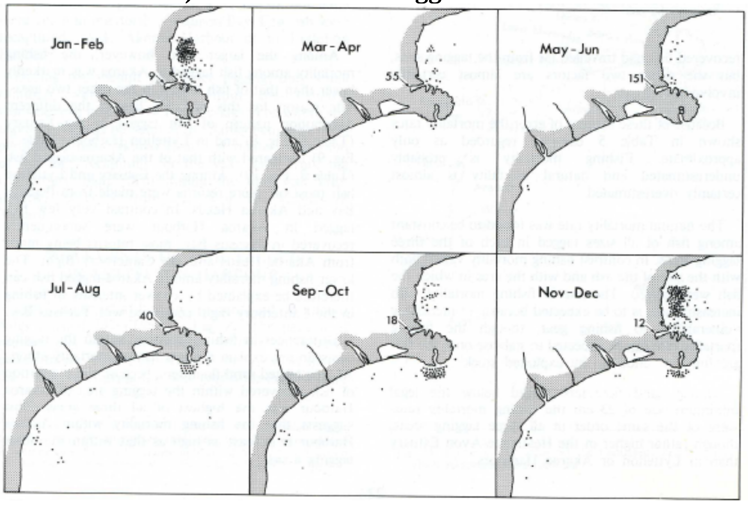


Figure 33: (left) Estuaries in which sand flounders were tagged – Avon-Heathcote (5 767 fish), Lyttelton (3 390 fish), and Akaroa (5 894 fish); (right) Juvenile sand flounder, sampled in Kaipara Harbour. (Sources: a) Coleman (1978), b) Crispin Middleton, NIWA.)

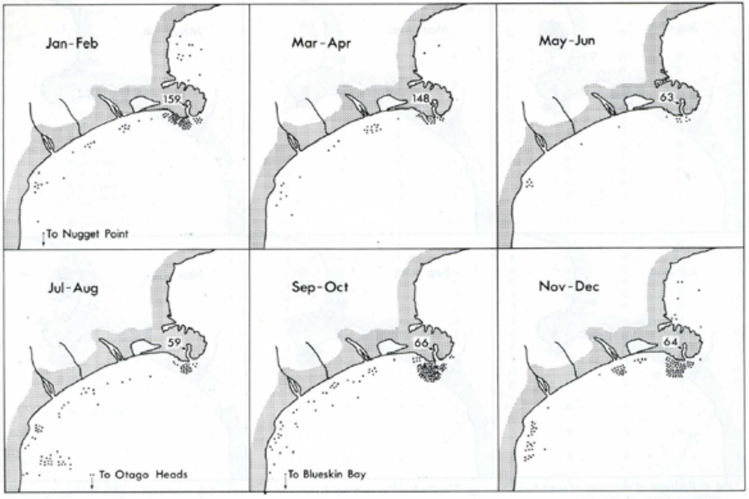
Most flounders left the estuarine tagging areas within a year of being tagged. Increasing returns from coastal areas, in particular Pegasus Bay and the waters off Akaroa Head, occurred from August to November, and this was taken to be when about most of the flounder moved out of the estuaries. This was at an estimated 2 years of age (Mundy 1968), at about 23 cm for females, and 'rather less' for males (Colman 1978). Few tagged fish were caught in the estuaries above this size. Tagged flounder from the Avon-Heathcote and Lyttelton estuaries were found in the same coastal areas, both north and south of Banks Peninsula (Figure 34). Tagged flounder from Akaroa estuary displayed a different movement dynamic, with most fish being recaptured in the Canterbury Bight rather than Pegasus Bay. A patch known as the "Winter Ground", 15–20 km southeast from Timaru, is a known spawning ground for sand flounders, while a second area in 30–45 m water depth off Akaroa Heads is thought to be a second (Colman 1978). Tagged fish from all three estuaries were found together during the spawning season. It was suggested that the general southward movement of sand flounders off the east coast of the South Island, also noted by Tunbridge (1966), probably had the effect of maintaining the flounder populations over the same area of coast, as planktonic eggs and larvae would be carried north by the Southland Current (Heath 1972). This would result in fish moving south to spawn in the Southland Bight providing eggs and larvae that would eventually metamorphose near Banks Peninsula and its nursery bays, while fish spawning near Akaroa Heads would provide eggs and larvae for settlement into Pegasus Bay and its estuaries (Colman 1978).



a) Avon-Heathcote tagged flounder



b) Lyttelton Harbour tagged flounder



c) Akaroa Harbour tagged flounder

Figure 34: Positions of recapture of sand flounders tagged in the Avon-Heathcote, Lyttelton and Akaroa estuaries, and subsequently recovered outside the tagging area at different times of the year. Numbers recaptured inside the estuary are given as a number. (Source: figures 8, 9 and 10, from Colman (1978).)

This work very clearly linked adult sand flounder populations to three estuarine nursery grounds. Similar spatial patterns of high juvenile sand flounder abundances, but low numbers of adults, are also seen in northern estuaries (e.g., Francis et al. 2005), and similar movement patterns are thought to occur. It would appear that increases in mud content from sedimentation, including an expansion in the spatial extent of such habitats, might actually be advantageous for this species. However, it should be noted that within such habitats, we do not have a clear understanding of what environmental conditions are important to juvenile flounders (including yellow-belly as well as sand flounder), and it may yet prove that land-based run-off into these areas, including contaminants, are deleterious at both the individual fish and wider population levels. One-off sampling of these three southern harbours in 2005 found relatively low numbers of sand flounders (Francis & Morrison, unpubl. data), and we have little understanding of the current status of these estuaries as habitat for sand flounders, more than 40 years after the tagging study was completed. Current changes to the disposal of Christchurch sewage into the Avon-Heathcote Estuary is likely to produce large changes in the functioning of this system.

8. RELATIVE SCALE OF LAND-BASED IMPACTS ON FISHERIES AND BIODIVERSITY

With our present knowledge of land-based impacts on fisheries in New Zealand being rather modest, it is premature to quantify their scale of impact versus other stressors such as recreational, and especially commercial, fishing. As commented previously, negative trends in estuarine and coastal fish population sizes and associated fisheries catches over the last 100 years probably arise both from over-fishing and land-based degradation of important components of the underpinning ecosystem. It is not currently possible to tease out the relative magnitude of these past effects, and for most species and systems we will probably never be able to do so. However, broadly speaking, we would expect to see greater effects on species that are found in those areas (and associated habitats) most vulnerable to land-based impacts, such as estuaries, sheltered coastal embayments, and where large rivers empty directly onto the coast. These may include species that spend most or all of their life cycles in such areas (e.g., shellfish such as cockles, pipi, and to a lesser extent paua, kina, and scallops), or that have nursery phases in such areas, depending on the context of the wider ecosystem (e.g., snapper, trevally, grey mullet, short-finned eels, and tarakihi in more structurally complex habitat elements, and sand and yellow-belly flounder on less structured sand and mud substrates).

However, notwithstanding quite large quantitative data gaps, we would suggest that impacts have in fact been quite profound, based on present-day observations of some species, habitats, and systems. More obvious ones may include the substantial loss of seagrass meadows (especially subtidal elements) from wider regions where they are known to support significant finfish nursery functions (e.g., from Manukau, Whangarei, Waitemata, and Tauranga Harbours), and reductions in the availability of harvested species such as cockles and pipis from areas that are not thought to have been overfished. As also noted by Airoldi & Beck (2007) for European systems, many other habitats (especially in the subtidal) may have been subject to very substantial losses from human-driven impacts, but the available information is scattered and anecdotal, if any indeed exists at all.

9. GAP ANALYSIS AND SUGGESTIONS

Our current understanding of land-based impacts on coastal fisheries and supporting biodiversity in New Zealand is very limited. Where specifically to investigate these impacts around New Zealand is open to debate, and we would suggest that some form of comparative environmental gradient approach would be ideal. This would include both estuaries and coastal areas, with and without significant river inputs from agricultural or more pristine catchments. Specific coastal areas falling along this gradient might include: East Northland with its complex coastal topography of rivers, estuaries, embayments and islands; East Cape region with its high sediment loads; the Marlborough Sounds with its complexity of sheltered and exposed seascapes and associated biogenic habitats; and Foveaux Strait as a relatively oceanic system. Specific estuarine systems that might be targeted could include the Kaipara Harbour (a key nursery for a number of west coast North Island fish stocks), Parengarenga and Ranguanu harbours on the east Northland coast as known high value fish nurseries for more northern fish species, and more southern estuaries such as Bluff/Awarua, and Paterson Inlet and others on Stewart Island, as suspected nursery grounds for more southern fish species. Regardless of the specific geographic areas that might be selected, there are some broad generic research questions to be addressed. These should be researched in combination with an examination of the relative role of marine-based stressors, such as fishing. Suggestions include the following:

- FISH–HABITAT ASSOCIATIONS: Fundamental and systematic inventorying of what fisheries species are associated with what habitats (and habitat elements) across different life stages, at both regional and national scales. This needs to incorporate biogeographic factors such as coast and latitude, as well as how the landscape context of a given area of habitat affects its functions. Special attention should also be paid to the possibility of habitat 'bottlenecks", which may limit the overall number of fish able to be produced by a given system. Currently we have little understanding of these relationships (from both the point of view of individual fished species, and specific habitats) making the basic assessment of fisheries habitat values, and potential threats to these values, problematic for many species (though not all).

- HABITAT CONNECTIVITY: A better understanding of the spatial linkages via movement between different habitats and areas as fished species pass through different life stages. Impacts in relatively limited habitats/areas may have cascades of effects into far distant locations: a good documented example is Kaipara Harbour snapper nurseries linkages to the west coast North Island ecosystem. Without accounting for such connectivity, we will always be limited in our ability to identify the important factors driving variation in harvested species abundance in a given area, and where management efforts might best be directed.

- RIVER PLUMES: A better understanding of how river plumes influence coastal fisheries, both positively and negatively. This should incorporate the different types of marine settings rivers empty into (estuarine, sheltered coast, or exposed coast). Changes in river flows and associated debris and nutrient loads could also be incorporated into this work, to assess how changes in water extraction might interact with coastal fisheries.

- LAND-BASED STRESSOR IMPACTS: The actual effects of sedimentation and eutrophication on selected fished species, fisheries habitats, and habitat landscapes. These include both direct impacts, such as adverse physiological and behavioural effects on fished species, and indirect impacts such as loss of critical habitats, and reductions in prey assemblages. The potential effects of eutrophication on coastal fisheries have been unexplored in New Zealand, and deserve attention. Ideally, species and areas should be selected within a framework which will allow for the findings to be extrapolated to other similar species and areas.

- INTEGRATION WITH MARINE STRESSOR IMPACTS: Stressors do not operate in isolation. Marine based stressors (e.g., fishing, mining, and dredging) and land-based stressors will interact with each other, with their relative importance at a given location depending on the distance to the source of the different stressors, and what natural systems and processes are operating at that location. For instance, more ocean-influenced systems that have seldom experienced land-based influences may have stronger responses to such influences when they do occur, than more land influenced systems that have evolved under continual inputs (e.g., sediment inputs from the Southern Alps).

- SPATIAL MAPPING AND SYNTHESIS: Such thinking also ultimately lends itself, in a management sense, towards the spatial zoning of marine ecosystems (based on functions and stressors), and how then to regulate human activities and impacts relative to these different zones. GIS and other technologies are available that make such synthesis possible. All field surveys, and associated experimental work, should be spatially explicit so that outcomes can be incorporated into GIS frameworks, both as decision support management systems and as research tools that can help direct and interpret new research initiatives.

10. CURRENT AND PROPOSED RESEARCH PROJECTS

A range of relevant research programmes by different research groups and providers is currently under way. We list here those of which we have some knowledge, but there will be additional ones not listed.

- "Marine Recreation" project: This FRST funded work is quantifying the interactions between recreational snapper fisheries, fish populations, and their underpinning habitats in the inner Hauraki Gulf. While the direct focus of the programme is on the effects of recreational fishing, the information being collected (habitat mapping, fish-habitat associations, fish movement, recreational fishing patterns and catch, biogenic habitat distribution and abundance) is also relevant to land-based stressors, which also affect these features and processes. For example, Tamaki Strait has had substantial inputs from the land, especially fine sediments, as well as multiple invasions of non-indigenous species, including bivalves, ascidians, and polychaetes. Collectively, these multiple stressors affect the functioning of this ecosystem. All data sources and derivatives are being integrated in the GIS "RECFISH" (see Morrison et al. 2008). Project Code: CO1X0506, Contact NIWA.

- "Tamaki Strait Tier II monitoring project": A two year ARC monitoring programme is providing a broad description of the habitats and species that occur in the Tamaki Strait. The programme is designed so that in future decades any large-scale changes to this system will be able to be detected. Subtidal and intertidal areas are surveyed. The methods used are a mix of broad scale survey techniques such as side-scan sonar, video surveys (remote and diver operated), and grab samples. Finer resolution sampling of epifauna, infauna, and flora is also carried out in selected locations using transects and quadrats for both rocky and soft sediment areas. Contact: Auckland Regional Council.

- "Biogenic habitats as areas of particular significance for fisheries management": A desktop information review, with a focus ranging from the intertidal out to sea-mounts within the Exclusive Economic Zone (EEZ). Particular focus is being directed at biogenic habitats role as nurseries, and as spawning grounds, and an assessment of threats to these values. Project Code HAB200701. Contact: Ministry of Fisheries.

- "The ecological and environmental services of *Perna canaliculus* in soft sediment systems": This MSc project is looking at the role that green-lipped mussels play in soft sediment systems, including as habitat for fish and invertebrates, as biodiversity and animal productivity enhancers, and as filterers of the water column. This work encompasses both field surveys and experiments. A current experiment is looking at possible bottlenecks to the re-establishment of mussels in the Firth of Thames (e.g., effects of larval supply, settlement surfaces, and the presence of adult con-specifics for successful settlement/recruitment; growth and survival of transplanted adult mussels). Contact: Leigh Marine Laboratory; NIWA, Auckland; Auckland Regional Council.

- "Taking stock: effects of climate variation and human impacts on New Zealand's marine ecosystem over the past millennium": This Biodiversity Fund project is working to reconstruct what New Zealand's marine ecosystems looked like before the arrival of humans. Numerous and diverse information sources are being used, and include the assessment of significant changes in the marine environment resulting from human impacts. Project Code ZBD200505. Contact: Ministry of Fisheries.

- "Rocky reef ecosystems – how do they function? Integrating the roles of primary and secondary production, biodiversity, and connectivity across coastal habitat landscapes": This Biodiversity Fund programme is looking at a gradient of reefs in the Greater Hauraki Gulf and beyond (East Coast Bays to the Poor Knights Islands), including four Marine Protected Areas (MPA). Objectives include an assessment of assemblage structuring dynamics (e.g., predation versus sedimentation, in situ versus external primary production), connectivity of key finfish species (snapper, parore), and the building of qualitative interaction models. Project Code: ZBD200509. Contact: Ministry of Fisheries.

- "Research on biogenic habitat-forming biota and their functional role in maintaining biodiversity in the inshore region (5– ~150 m depths)": This Biodiversity Fund project aims to identify significant areas of biogenic habitat on the shelf (5–200 m water depth) through interviews with fishers, researchers and others with relevant knowledge, following by targeted sampling using multi-beam sonar and a deep towed imaging system (DTIS), deployed at night to count sleeping fish. These findings will be incorporated into models of where such habitats occur, their ecological and fisheries roles, threats to these, and what might be done to mitigate these threats. Project Code: ZBD200801. Contact: Ministry of Fisheries.

- "Habitats of particular significance for fisheries management: Kaipara Harbour": This project, currently in the tendering phase, has an overall objective "To identify and map areas and habitats of particular significance in the Kaipara Harbour that support fisheries, and assess potential fishing and land-based threats to their function". Specific objectives are, a) collate and review information on the role and spatial distribution of habitats in the Kaipara Harbour that support fisheries production, b) assess historical, current, and potential anthropogenic threats to these habitats that could affect fisheries values, including fishing and land-based threats, and c) design and implement cost-effective habitat mapping and monitoring surveys of habitats of particular significance for fisheries management in the Kaipara Harbour. Project Code: ENV2009/07. Contact: Ministry of Fisheries.

- "Integrated Catchment Management for the Motueka River: From ridge tops to the sea". This research programme is assessing the Motueka River catchment and how it functions, and its associated marine influence in Tasman Bay. An extensive website is available for this programme, with numerous documents and presentations available (http://icm.landcareresearch.co.nz/). Contact: Cawthron Institute.

- "Review of Challenger (SCA 7) scallop fishery": The objective of this project is to assess factors that may have affected the functioning of the Challenger scallop fishery (Golden Bay, Tasman Bay, and the Marlborough Sounds). Using the information on various biological, physical, and fishing-related factors, including land-based influences, this review will evaluate the relative influence that each of these factors has likely had on scallop abundance, and possibly how some of these factors could be managed or mitigated in the future. Contact: Ministry of Fisheries.

- "Toheroa abundance": The objective of this desktop review is to investigate variations in the abundance of toheroa (*Paphies ventricosa*) by examining factors that influence recruitment and mortality. A wide range of scientific, customary, and historical information on toheroa populations is being acquired, analysed, and reviewed. It is envisaged this work will form the foundation on which to build future research on toheroa, with a review to restoring populations of this iconic New Zealand shellfish. Contact: Ministry of Fisheries

- "Coastal marine risk assessment": This project has two objectives: a) to collate existing information on the distribution, intensity, and frequency of anthropogenic disturbances in the coastal zone that could be used in a risk assessment model to estimate their likely aggregate effect on ecosystem function across habitats and over different scales of ecosystem functioning and biological organisation; and b) to develop a risk assessment framework in conjunction with a variety of stakeholders and environmental scientists. Project Code: BEN200705. Contact: Ministry of Fisheries.

11. LINKS, INFORMATION SOURCES, RESOURCES, CONTACTS ETC

A number of organisations have useful on-line resources. This list is not exhaustive.

- Auckland Regional Council: all the technical reports produced by the ARC that are cited in this report can be downloaded from http://www.arc.govt.nz/plans/technical-publications/technical-publications_home.cfm. Highlight documents include Gibbs & Hewitt (2004) (TP211)

- Environment Waikato: various publications can be downloaded from http://www.ew.govt.nz/Publications/New-Publications/. Highlight documents include Jones (2008) (TR2008/12).

- Environment Southland: various publications can be downloaded from http://www.es.govt.nz/Departments/EI/EIreporting.aspx?sm=i_a. Highlight documents include several listed under the "Environmental Monitoring" section (Waikawa Estuary and Waituna Lagoon reports)

- Environment Canterbury: in particular the erosion and sediment control guidelines at http://www.ecan.govt.nz/Our+Environment/Land/ErosionAndSediment/ErosionSedimentControlGuidelines.htm.

- The Ministry of Fisheries: while Aquatic Environment and Biodiversity Reports (AEBR) and Marine Biodiversity and Biosecurity Reports (MBBR) are not yet online, other useful documents that can be downloaded include Fisheries Assessment Reports (FARs) and Fisheries Assessment Research Documents (FARDs) from http://fpcs.fish.govt.nz/science/ResearchDocuments.aspx, which report on various fisheries research investigations; and plenary documents (which summarise the current status of different fisheries stocks, at http://fpcs.fish.govt.nz/Science/Plenary.aspx.

- As a useful example of how overseas workers have thought about the interactions of multiple stressors on coastal fisheries, we highly recommend the work by Buchsbaum, J.; Pederson, W.E.; Robinson (eds). (2005). The decline of fisheries resources in New England: evaluating the impact of overfishing, contamination, and habitat degradation. MIT Sea Grant College Program, Cambridge, MA, MITSG 05–5. Available at: http://massbay.mit.edu/publications/NEFishResources/index.html.

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