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Land-based effects on coastal fisheries and kaimoana and their habitats – a review

New Zealand Aquatic Environment and Biodiversity Report No. 309

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EXECUTIVE SUMMARY

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In 2009, a desktop literature review was completed for the (then) Ministry of Fisheries, on land-based effects on coastal fisheries and supporting biodiversity. Since that time new research and management initiatives have advanced this area of knowledge, both in New Zealand and internationally. This current review updates and expands on that previous review. Much of the material from the 2009 review is retained, as the fundamental processes and supporting body of knowledge remains directly relevant.

Land-based effects on coastal fisheries may occur through a diversity of mechanisms. Changing inputs from the land have included increased 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 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 targets, under which stocks are fished down to a level where productivity is thought to be highest. It is assumed 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 are highly likely to have changed (reduced) the carrying capacity of the environment, including for valued fisheries and kaimoana species and assemblages. 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, mussels), reductions in settlement success and survival of larval and juvenile phases (e.g., pāua, kina), and reductions in the foraging abilities of finfish (e.g., juvenile snapper, grey mullet). Indirect effects include the modification or loss of important nursery habitats, especially habitat-forming (biogenic) species.

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 seafloor, 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. Little work has yet been done on the potential impact of

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eutrophication on coastal fisheries in New Zealand, though it may be modest relative to other areas of the world. The Firth of Thames is one area of higher concern for increasing nutrient levels over time.

The value of Submarine Groundwater Discharge (SGD), particularly the freshwater component, to coastal systems is being increasingly recognised. SGD waters often have cooler temperatures, lower salinity, and higher dissolved nutrient levels than the marine waters that they discharge into, through seafloor features such as seeps and springs). Studies on these seafloor features internationally are revealing hotspots of higher primary and secondary productivity, including fished species valued by humans. With water supply being an increasingly important issue, and associated water extraction for human use fundamentally increasing (both by drawing from aquifers and from surface waters that ultimately recharge many aquifers), new downstream issues are likely to emerge in the marine environment.

Land-based impacts are ideally managed at their source (within land catchments). In this review we collate and integrate the information on land-based stressors available at the regional and national scale in New Zealand, with a key focus on sediment and nutrients. We also summarise what intervention measures have been implemented in the New Zealand catchments that have attempted to mitigate the impact intensity of these land-based stressors.

Long-term, multi-generational observations and experiences of land-based impacts on the marine environment are a key knowledge source. In this review we collate the accessible information on the effects of land-based stressors on kaimoana, as expressed by iwi and hapū around Aotearoa New Zealand. The key purpose is to provide background into some of the concerns and interests held by selected iwi and hapū around land-based stressors and their impacts on kaimoana, particularly due to sediment.

We suggest that fundamental gaps remain in our knowledge of how land-based stressors affect coastal fisheries, both in New Zealand and globally. 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 are made for research that would advance our ability to understand and best manage these issues. 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., pāua *Haliotis iris*, kina *Evechinus chloroticus*, rock lobster *Jasus edwardsii*), and on intertidal soft shores (e.g., cockles *Austrovenus stutchburyi*, pipi *Paphies australis*). 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 timescale, 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 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 timescales 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 review, we update and expand on the Morrison et al. (2009) review, which covered the current state of knowledge as of 2009, on 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. We also discuss three areas not covered previously: an overview of impacts on Māori kaimoana as expressed by iwi and hapū; the emerging potential role of groundwater delivery with associated higher nutrient levels into the coastal environment; and the available data and models for estimating sediment and nutrient outputs from New Zealand's land catchments.

1.1 Objectives

Overall Objective:

To provide an updated state of knowledge of land-based effects on coastal habitats and the fisheries and kaimoana species they support.

Specific Objectives:

1. To review current state of knowledge of land-based effects on coastal fisheries and kaimoana species in New Zealand, and important developments in this research area internationally.

2. To collate information on land-based stressors to coastal ecosystems, mitigation measures taken, and to provide a detailed list of all available datasets and spatial layers at regional and national scales

1.2 Scope and limitations of review

This review updates the Morrison et al. (2009) report. At the direction of Fisheries New Zealand, we have retained much of that previous review in this update, as much of the wider background remains largely unchanged (e.g., the processes that generate land-based inputs), while for ecological processes, the new work adds to the research foundations/findings previously created. 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. We deliberately and explicitly focused 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 focus was on coastal fisheries. For example, while sedimentation strongly affects soft sediment benthic assemblages in general, we specifically focused 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 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.

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, many 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 Hawke's Bay

(Tūtira 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 Hawke's Bay (Page et al. 1994). Clearance of this region started about *c* 800–500 years ago (Wilmshurst et al. 1997).

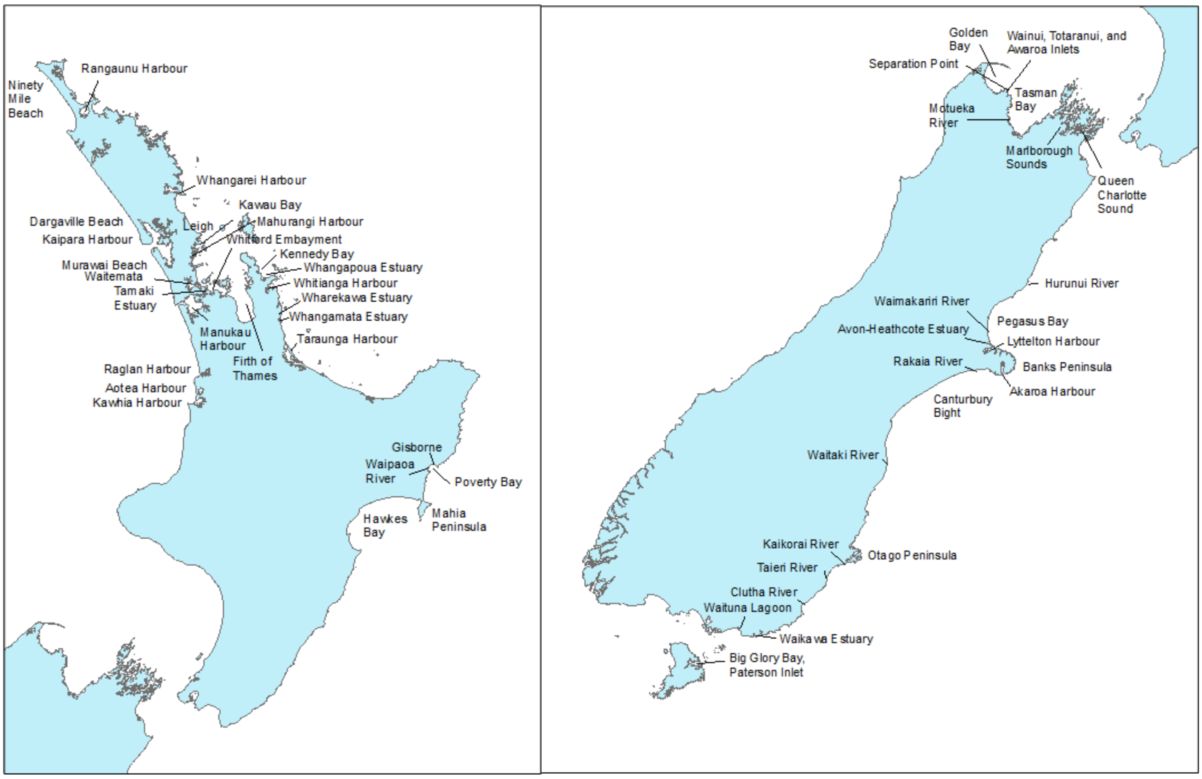

Figure 1: Locations of place names mentioned in the text, for the North Island and South Island, respectively.

2.2 The arrival of humans

Following the arrival of Māori, around half of New Zealand's vegetation cover was removed by burning (Poole & Adams 1994). At this time, most of the land used by Māori 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 landslides (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, at almost 1% of total world sediment yields (Robertson & Stevens 2006).

Figure 2: Remnant of beech 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 Hawke's 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 shorelines) 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. Off the west coast of the South Island, rivers deliver about 12–26 million tonnes of sediment a year, resulting in a sedimentation rate on the shelf estimated to average 1–2 mm y^{-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 Waitōtara, Whanganui, Rangitīkei, and Manawatū rivers contributes *c* 4.44 million tonnes y^{-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.

3. PHYSICAL CONNECTIONS BETWEEN LAND-BASED ACTIVITIES AND 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 focused 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 (Buchsbaum 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 accelerates 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 3) are vulnerable to dry summers and, 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 3: 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 4). These occur on all land types and are caused by intense rainfall events. Landslide failure rates increase with slope; most commonly occur 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 the occurrence of landslides) during large storm events (Jones 2008).

Figure 4: Soil-slip erosion on hill-country pasture. Domestic deer are present on the foothill on the left side of the image. (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 & Marsden 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, and nutrient concentrations, and higher water clarities, than those draining pine forest and pasture (Quinn & Stroud 2002).

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 kānuka (*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 & Marsden 1999). Sediment is generated both during and following forest harvesting (Jones 2008) (Figure 5) (see also Whangapoua Estuary Section 6.1).

Figure 5: 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. A number of these are listed in Appendix A. While these models provide a means of estimating sediment yields or loads it should be noted that 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 and oceanography. 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 timescales 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 y^{-1}, rising to 3.0–7.2 mm y^{-1} during catchment deforestation (1880–1945), and 5.0–8.0 mm y^{-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 y^{-1}, increasing to 11 mm y^{-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 1985); Whangapoua Estuary, with pre-Polynesian SAR rates of 0.03–0.08 mm y^{-1}, increasing to 0.12–0.13 mm y^{-1} following Māori occupation, and to 0.89–1.5 mm y^{-1} following European forest clearances.

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

Region	Land-use practice	Pre-Polynesian SAR (mm y^{-1})	Post-Polynesian SAR (mm y^{-1})	Post-European SAR (mm y^{-1})	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 (1994)
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. (2005a)
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)
Waitematā Harbour	Deforestation	<1.5	2.5 (700–1100 BP)	3 (1841–present)	Hume & McGlone (1986)
Pāuatahanui Inlet	Deforestation	1		2.4–3.4 (after 1850s), 4.6 (since mid 1980s)	Swales et al. (2005b)
Wainui, Tōtaranui, and Awaroa inlets	Deforestation	0.5–1.7	1.62–2.7	2.3–3.3 (within last 30 years)	Goff & Chague-Goff (1999)
Waikawa Estuary	Deforestation		1.5 (1878–1967)	3.1–10.7 (1967–2007)	Robertson & Stevens (2007)

On the west coast, harbour-wide, Raglan Harbour SAR has averaged 0.3–0.5 mm y^{-1} over the last 8000–6500 years (Swales et al. 2005a). 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 y^{-1} since 1890. Pine pollen presence suggested that the SAR rate has further increased to 2.5 mm y^{-1} since the early 1990s, with a maximum of 8 mm y^{-1} at a site in Okete Bay (Swales et al. 2005a). 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 Māori 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. 2005a), 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 y^{-1}, when the surrounding catchments were vegetated in podocarp hardwood forests. Following Māori settlement and associated forest clearance, SAR rates increased to 2.4 mm y^{-1}, and, following European land clearances from about 1840 onwards, SAR increased to 6.25 mm y^{-1}, with significant increases of heavy metals (Cd, Cu, Pb, and Zn) in the most recent layers (Abrahim 2005). In the Papakura Estuary, pre-human SAR rates were 0.2–0.5 mm y^{-1}; these rates increased three-fold to 0.8–1.6 mm y^{-1} following European forest clearance and subsequent agriculture in the mid 1800s and at the top of the estuary have averaged 32.6 mm y^{-1} since 1960 (Swales et al. 2002). In the Mahurangi Harbour, following catchment deforestation (1850–1900), 3 metres of sediment have 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 generated from nine floods from 1953 to 1995. In Lucas Creek, in the upper Waitematā Harbour, rates increased from less than 1.5 mm y^{-1} before human arrival, to 2.5 mm y^{-1} during Polynesian forest clearance (700–110 BP), and then to 3 mm y^{-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, Pāuatahanui Inlet, sediment cores returned SAR estimates of about 1 mm y^{-1} over the past several thousand years, increasing to about 2.4 mm y^{-1} over the last 150 years, with a further increase to about 4.6 mm y^{-1} from the mid 1980s to the early 2000s (Swales et al. 2005b). In the Horokiri subcatchment, the sub-catchment's size and steepness, combined with large-scale planting of pine forest since the 1970s, resulted in a SAR of 10 mm y^{-1} to the early 2000s (twice the rate of elsewhere in the Pāuatahanui Inlet) (Swales et al. 2005b). Harvesting of the 800 hectares of pine forest to around 2020 was proposed to likely further increase this sedimentation rate (Swales et al. 2005b). At the top of the South Island, coring in the Wainui, Tōtaranui, and Awaroa inlets, inside the Abel Tasman National Park, Goff & Chague-Goff (1999) quantified a 1700 y 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 y^{-1}, which increased following their arrival to 1.62–2.7 mm y^{-1}, increasing in the 1960s to the 1990s to 2.3–3.3 mm y^{-1} Goff & Chague-Goff (1999). 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 about 145 km south of Dunedin, the average SAR rate from 1878 to 1967 was 1.5 mm y^{-1}, increasing to 3.1 mm y^{-1} from 1967 to 1996, and to 10.7 mm y^{-1} from 1996 to 2007 (Robertson & Stevens 2007). Half of the estuary 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).

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 to 0.6 mm y^{-1} (7000 BP to 1960) to 2.8 mm y^{-1} since 1960 (Cadmus & Schallenburg, unpublished data, referenced by Ryder Consulting 2008).

Direct run-off to the sea

In some areas, slope run-off carrying sediment may flow directly into adjacent coast zones (Figure 6, Figure 7). 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 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 6: Aerial photograph of Māhia Peninsula, from the west, showing suspended sediment fringe around the land, taken 15 April 2005. (Source: Anna Madarasz-Smith, Hawke's 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} y^{-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} y^{-1}), which could increase to 7000 t (218 t km^{-2} y^{-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 7: 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 (Figure 8, Figure 9). 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 Rākaia rivers deposits 100–200 km north on the up-drift side of Banks Peninsula (Gibb & Adams 1982).

Figure 8: 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 Kāwhia 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 9: 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, and stable carbon and nitrogen isotope signatures in bivalve shellfish) found an influence extending out at least 6 km from the mouth, with the strongest influences in the first 2 km (Forrest et al. 2007). Sites within 2 km 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. Forrest et al. (2007) 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.

Cornelisen et al. (2011) examined the Motueka River's delivery of faecal contaminants into Tasman Bay during a moderate flood event (flows of up to 400 cubic metres a second). A shallow low-salinity plume extended at least 6 km into the bay, including the region's biggest Aquaculture Management

Area. Faecal contamination was found in both water and mussel samples, derived from ruminant animal livestock (cows and sheep). Monitoring buoys 3 and 6 km offshore of the river mouth detected the persistence of the river plume for a week after the flood event. Salinity was reduced at 3 m water depth (where the buoy sensors were located), as was water temperature. Light attenuation (loss) was considerably higher post-storm than pre-storm, and, for at least a week, the amount of light at 3 m on a sunny day was half the level recorded prior to the arrival of the river plume. A one day boat-based survey two days after the storm event found a suspended sediment layer in the upper few metres of the water column (less than 3 m water depth), which had only a marginal impact on the slowly decreasing turbidity levels at 3 m depth (as measured by the buoys).

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 mm 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 landslides 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 y^{-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 5836 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 m 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 1983). 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 timescale, 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 10 for an example of current day sediment plumes).

Figure 10: 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), *Escherichia 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. Complementing the summary given here, subsequent sections describe national and regional scale freshwater monitoring networks and recent analyses of data that further demonstrate the influence of land use on freshwater bodies.

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 focused 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 (now Auckland Council), which consists of a comprehensive water quality monitoring network across 27 estuary and near-shore sites, from 1987 to 2008 (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 Māngere Wastewater Treatment Plant. Following the decommissioning of the Māngere oxidation ponds in 2002, dramatic improvements have occurred, especially in levels of ammoniacal nitrogen, total phosphorus, and suspended sediments.

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 Waitākere, Auckland, and Manukau cities i.e., Henderson Creek to Cox's Bay along the southern shores of the Waitematā Harbour; the upper reaches and side-branches of Tamaki Estuary; and Māngere 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, except for Māngere 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 Ōrewa 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 several upper Waitematā sites, while lead and zinc concentrations were below TEL thresholds except at Hellyer's Creek. Zinc concentrations were increasing rapidly in Lucas Creek and would likely 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 levels of contamination (or a covariate of copper, lead, and zinc) were 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 (now Greater Wellington), across 17 sites in Wellington Harbour, 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 shellfish 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 Submarine Groundwater Discharge (SGD)

An emerging new area of interest is the role of submarine groundwater discharge (SGD) to the coastal environment and the potential for human land-based activities to result in changes to the characteristics of SGDs of significance for coastal ecosystems. SGD is defined as "*any and all flow of water on continental margins*" (Burnett et al. 2003, Moore 2010), but the key interest from a biological point of view is the freshwater/terrestrial fraction (Moosdorf & Oehler 2017). Globally, an estimated 90% of total SGD is recirculated seawater and 10% freshwater (Kwon et al. 2014). That freshwater SGD is estimated to contribute 0.01–10% of the total freshwater volume that arrives at the coast, with the relative contribution varying more widely at local scales depending on the presence of larger rivers (Garrels & Mackenzie 1971, Zektser & Loaiciga 1993, Church 1996, Taniguchi et al. 2002). Research on SGD has been relatively limited until the last decade or so due to the difficulty in finding and measuring SGD features (Taniguchi et al. 2002). SGD may be both volumetrically and chemically important to coastal water and chemical budgets (Johannes 1980). The geochemical cycles of some major/minor elements such as nitrate and phosphate arriving in the coastal zone can be strongly influenced by direct freshwater discharge and/or chemical reactions in seawater being recirculated through shallow coastal aquifer systems (COSOD II 1987, Buddemeier 1996). The presence of submarine springs and seeps may be a significant pathway for dissolved substances and diffuse pollution to arrive at the coast (Burnett et al. 1996, Burnett et al. 2001).

The two main factors that determine SGD rates are a) driving force, and b) the development of river systems. Driving force is created by a combination of hydraulic connectivity (the geology of a region) and the groundwater discharge rate (the balance of precipitation, evapotranspiration by vegetation). River development is controlled by geology and topography. How SDG is measured and estimated is beyond the scope of this review. Taniguchi et al. (2002) and Moore (2010) provide good overviews of the different approaches to detect and measure SDG, at a range of spatial scales. Moosdorf & Oehler (2017) provide an engaging review of the societal use of fresh submarine groundwater discharge, extending back into history. Saltwater intrusions into coastal freshwater aquifers is a complementary process to SGD and has becoming an increasing issue as water extraction from aquifers by humans increases. Which dominates is a balance between hydraulic and density gradients in groundwater versus seawater, along a transect perpendicular to the shoreline (Taniguchi et al. 2002). Changing sea levels influence this balance.

Terminology related to SGDs are Offshore Freshened Groundwater (OFG) systems, which are very large offshore bodies of fresh and moderately brackish groundwater (concentration of total dissolved solids of <10 g l^{-1}) (Post et al. 2013). These can occur up to 100 km offshore of modern shorelines and to depths of 4.5 km below the seafloor surface. Most occur in shallow (< 300 m), poorly consolidated, clastic sediments in seawater depths less than 50 $m1^{-3}$, with the majority of OFG discoveries along passive continental margins, mostly in the Atlantic US and European margins. This distribution at least partially reflects various countries financial and scientific abilities to look for such OFGs.

Work in New Zealand on SGD is quite limited. Ridgeway & Stanton (1977) looked at water column and seafloor temperature and salinity patterns in Hawke's Bay, sampling across a 5-mile resolution grid. Stations were also targeted at four freshwater springs as marked on the navigation chart at the time (N.Z. 56, Hydrographic Branch) at around 50 km offshore, in 80 m water depth. No evidence of their existence was found, with only high water salinities found at the seafloor.

Williams (1977) examined the dynamics of the Te Waikoropupū Springs, found near Tākaka, in the Tasman Bay catchment. These are New Zealand's largest springs, with an average discharge of around 14 m^3 s^{-1}. Three main sites contribute, all within a 130 m radius of each other, and at 14–17 m above sea level. These springs are described as karstic, tidal, brackish, and artesian in nature. Their mean flow-through time is three to four years, based on tritium data. Their principal source is the upper Tākaka River, that sinks into its gravel bed 16–18 km south of the springs, at 46–58 metres above sea level. The springs are slightly saline, which indicates a 'subterranean marine connection', supported by the existence of submarine springs 1–5 km offshore. The connection process is suggested to be through a venturi-effect mixing of fresh and marine water, as swiftly moving freshwater currents in underground passages draw in seawater. The main spring has a salt to freshwater ratio of 1:200. The overall total groundwater system inclusive of the springs is 34 km, from the most distant inland input, to the offshore submarine springs (Williams 1977).

Further north in the Wellington region, a large aquifer also extends out and under Wellington Harbour, where freshwater springs are present at several seafloor locations (the Waiwhetu Artesian Aquifer). The springs in Wellington Harbour issue from an unconsolidated gravel aquifer. Leakage occurs from depressions in the harbour floor where the confining aquiclude (geological barriers to water flow) has been removed or weakened but is concentrated at a number of small vents in the floor of the depression through which leakage occurs. These vents are irregularly spaced and are occasionally on the sides of the depressions rather than on the base. Harding (2000) worked on the aquifer and its seafloor springs and provides a good overview of the system and marine investigations of it as part of harbour infrastructure operations over the last century (to 2000). He investigated thirteen zones in the harbour, with eleven of these holding significant seafloor depressions (Figure 11). Five of these zones '*exhibited artesian leakage to a significant degree*'. Eleven seafloor depressions were mapped in detail, and these ranged from 53 to 369 metres in diameter (noting that they were irregular in shape) at the seafloor surface, reducing to 14 to 160 m at their base. Maximum depression depths ranged from 13.3 to 31.3 m, with maximum slope angles of 4° to 17.3°.

Six of these depressions were dived on and were visually described (albeit in very poor water clarity conditions) as being "*conical features with sloping sides that extend from the relatively flat harbour* *floor down to a flat base*". Within these larger depressions were smaller submarine spring vents, usually one to two metres in diameter, and forming "*mini depressions within the base of the main depression*". They were extremely soft in their substrate nature. It was not possible to count how many were present in any given depression as the very poor underwater visibility made it very difficult to find them. Other sites of likely SGD were also noted for the Wellington region. One of these was at Barrett Reef, where divers have reported a 'blurriness' in the water (J. Goff, 1997, pers. comm.; R. Williamson, 1997, pers. comm.). This blurriness is a feature at sites of artesian leakage, caused by the mixing of waters with different refractive properties. Harding (2000) suggested that, if there was freshwater leakage around Barrett Reef, it was more likely to be part of the deeper Mocra Gravel Aquifer, rather than the Waiwhetu Artesian Aquifer.

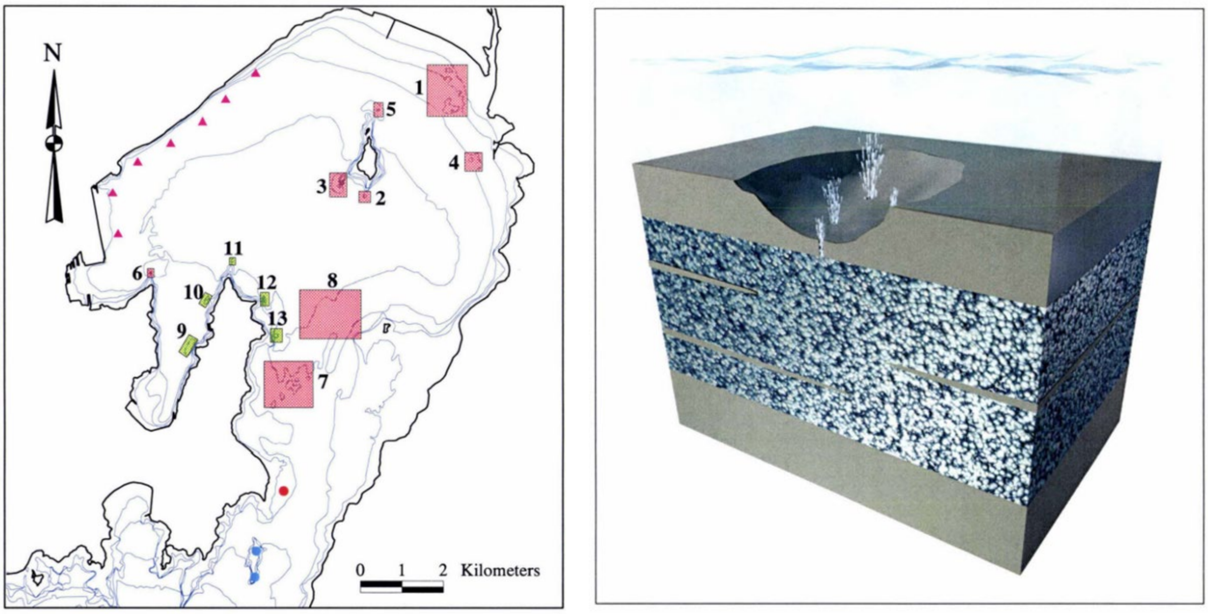

Figure 11: Left) Wellington Harbour study area of Harding (2000). Red regions represent areas that exhibit signs of present day (late 1990s) artesian leakage. Green regions represent other areas investigated which contained sea floor depressions, but where no signs of artesian leakage were found. Pink triangles show locations identified in the past as having artesian leakage, but not investigated as no sea floor depressions were present. Blue dots represent areas around Barrett Reef where SCUBA divers noted disturbance in the water indicative of artesian leakage. The dark red dot represents a suspected submarine spring discharge detected by NIWA side scan sonar in 1987. High resolution bathymetric surveys were undertaken in Zones I - 6. Bathymetric contour lines are shown in blue (5 m interval). (Source: figure 9.4.1.1. of Harding 2000.)

Right) Schematic diagram of a 'typical' submarine spring in Wellington Harbour. The blue, coarse section represents the Waiwhetu aquifer, the thin grey lenses within it represent overbank deposits. The thick units above and below the aquifer gravels represent the confining aquicludes, the Wilford Shell Beds (old coarse shell deposits below) and the Petone Marine Beds (fine marine sands above). Freshwater is depicted escaping from small vents around the base of the seafloor depression. (Source: figure 3.3.1 of Harding 2000.)

More recently, Wellington Water Limited (2018) undertook an exploration drilling project in Wellington Harbour to assess if the sub-harbour extension of the Waiwhetu Aquifer can provide an alternative resilient drinking water supply for central Wellington. They found that the submarine discharge was largely controlled by "*terrestrial groundwater conditions and stresses and is likely to be very responsive to changes in, for example, abstraction* [for human use]". Three discharge mechanisms were suggested: "*outflow through discreet submarine spring vents, widespread slow discharge across the Petone Marine silts, and outflow through the coarser grained sediment sequence in* [the] *harbour entrance area*". The total volume of offshore groundwater discharge was relatively low, as onshore (human) water abstraction took most of the Waiwhetu Aquifer throughflow. Previous modelling "*simulated an average* *offshore discharge of about 17,000 m^3 /day, of which 25-30% occurred in the harbour entrance area via vertical leakage and about 10% was from discreet spring vents*". This was described as a very small through-flow under the harbour. Furthermore, discharge rates were considered highly seasonal, with large summer season reductions from groundwater pressure drops due to reduced recharge (rain) and higher onshore water abstractions.

The full extent of Queen Charlotte Sound and the immediately adjacent coast has recently been fully mapped using multibeam sonar, along with ground-truthing using sediment grabs and dropped/towed video (Watson et al. 2020). The resulting bathymetric maps have revealed numerous pit/pockmark features on the seafloor (Figure 12). In inner Queen Charlotte Sound, 4181 pockmarks were mapped, which contributed 49% of all pockmarks seen across the overall mapped area. Most occurred in 25 to 50 m water depth, and collectively they covered 1.1 km^2 of seafloor. Two main types were seen (large and small), based on their size and spatial distribution. Large pockmarks had an average surface area of more than 1000 m^2 and were concentrated along the base of slopes immediately adjacent to the coastline (within 100–500 m). Small pockmarks were on average less than < 1000 m^2 and mostly found in flat regions in the embayments and main channel axis (Figure 12). For the large pockmarks, 172 were present, which contributed 29% of all large pockmarks mapped overall.

The central Queen Charlotte Sound area held 4234 pockmarks (50% of all pockmarks mapped), which mainly occurred in 30 to 60 m water depth and collectively covered 2.1 km^2 of seafloor. Of these, 395 were large pockmarks, which contributed 66% of all large pockmarks mapped overall. The spatial patterning of large and small pockmarks was very similar to that seen in the inner sound. The density per unit area of pockmarks in the central sound (36 km^{-2}) was around half of that seen in the inner sound (63 km^{-2}). Few if any pockmarks were seen in the outer Queen Charlotte Sound area (based on examination of figures 6 and 7 of Watson et al. 2020; numbers not given). In the Tory Channel region, only 60 pockmarks were present. These all occurred in less than 30 m water depth and were confined to within the shallow embayments flanking the Tory Channel proper.

Watson et al. (2020) suggested that the presence of both large and small pockmarks in Queen Charlotte Sound suggested multiple forms of fluid escape from the seafloor. As with other pockmarks observed overseas, they were considered most likely to be a product of seepage of shallow subsurface gas from biogenic/thermogenic reduction of organic matter (e.g., Scanlon & Knebel 1989, Hovland et al. 1992, Fleischer et al. 2001, Rogers et al. 2006) and/or submarine groundwater discharges (e.g., Whiticar & Werner 1981, Whiticar 2002). The present-day status (active, transient, relic) of the processes forming these pockmarks is unknown. It was suggested that the larger pockmarks may be a result of more sustained fluid flow from established fluid conduits (possibly via groundwater seepage), while the smaller pockmarks might be a consequence of transient/isolated fluid expulsion (e.g., of shallow subsurface gas).

Similar pockmarks have also been multibeam sonar mapped in the adjacent Pelorus Sound and parts of Admiralty Bay (not yet published; but see Seabed Habitat Maps - Overview (arcgis.com)).While the Marlborough Sounds drowned valley systems are likely to be at the denser end of the spectrum of 'seafloor pockmark fields' for New Zealand, these fields also occur elsewhere in New Zealand. For example, recent multibeam sonar mapping of the Waimate area just north of Coromandel Harbour (west Coromandel Peninsula) has revealed small fields of pockmarks adjacent to coastal islands (Morrison et al. in review).

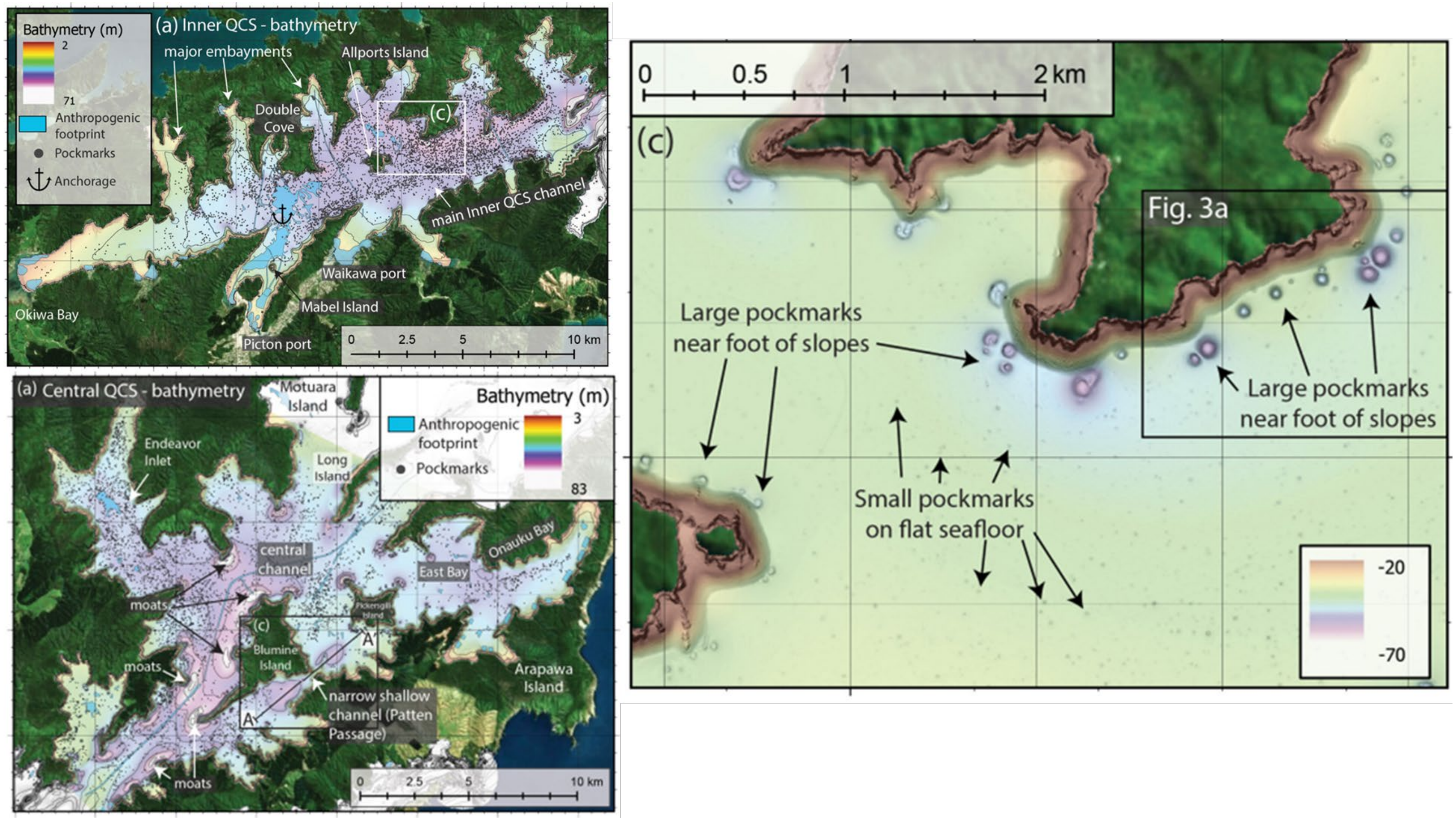

Figure 12: **Bathymetry and seafloor features of inner (upper left) and central (lower left) Queen Charlotte Sound. Pockmarks are shown as black dots, 10 m depth contour intervals as black lines, and seafloor regions affected by anthropogenic activities as blue polygons. Right) zoomed in view of area c from the inner sound, showing large pockmarks to scale. (Source: figures 2 and 5 of Watson et al. 2020.)**

Off the east coast of the South Island, Micallef et al. (2020) quantified an offshore freshened groundwater (OFG) system in the Canterbury Bight and characterised it as consisting of one main, and two smaller, low salinity groundwater bodies. The main body was found up to 60 km from the coast, at 110 m seafloor depth. The main OFG body extends up to 60 km from the coast to a water depth of 110 m, with a maximum thickness of at least 250 m and an estimated volume that ranges between 56 and 213 cubic kilometres. Input from rainfall on land and then active groundwater migration to offshore was inferred. However, the majority of the OFG was created from topographically-driven flow during sea level low stands in the last 300 000 years (i.e., created when sea levels were much lower, and the present day seafloor was above water and terrestrial). The presence of this OFG was originally inferred through a pore water salinity anomaly seen in a borehole driven down through the sediments in earlier general work (Expedition 317 Scientists 2011). Micaleff et al. (2020) searched for seafloor freshwater seepage across the Canterbury Bight but were unable to detect any. Some brackish water areas were found, but the authors were unable to differentiate between these being either localised diffused freshwater discharge, or alternatively changes in bottom water salinities due to currents.

3.6 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. The potential for impacts from changes from groundwater discharges is also possible (see above).

4. INFORMATION ON LAND-BASED STRESSORS

In this section we collate and integrate information on land-based stressors that are available at the regional and national scale in New Zealand. Our key focus is on sediment and nutrients. We also summarise what intervention measures have been implemented in the New Zealand catchments that have attempted to mitigate the impact of these land-based stressors. Appendix A supplements the description given here with a list of relevant datasets and spatial layers that have been developed at regional and national scales.

4.1 National and regional scale sources

4.1.1 Monitoring of land-based stressors

Water quality data are collected routinely at approximately 1150 river sites around New Zealand (Figure 13) primarily for monitoring the state and trend of water quality. The first major database to be developed was the National Rivers Water Quality Network (NRWQN) which started with 77 sites on 35 rivers in the 1990s (Smith et al. 1989). The NRWQN sites are reasonably well distributed over the North Island and South Island and drain about one half of the nation’s land area (Ballantine & Davies-Colley 2013). The 16 regional councils/unitary authorities also collect water quality data routinely as part of their State of the Environment (SoE) Monitoring Programmes. State of the environment reporting is a requirement of the Resource Management Act 1991 (RMA91). This SoE reporting is used to 1) detect changes in environmental conditions and monitor trends and determine their significance; and 2), to detect changes in environmental conditions following the implementation of council plans and strategies (LAWA 2021). Both the NRWQN and the regional council SoE monitoring programmes are based on regular monthly grab sampling of water quality samples. Such monitoring programmes tend to bias towards baseflow (i.e., low flow) and generally do not (or very infrequently) capture high flow conditions (Burt et al. 2011). Data from these sources is useful, particularly to provide information on water quality trends. However, because it is biased to baseflow conditions it is of limited use for calculating contaminant loads to receiving environments (such as the coast) (Snelder et al. 2017). This is because most contaminants are usually transported during high flow events and such infrequent sampling fails to adequately represent those conditions.

The selection of monitoring site locations has been somewhat arbitrary, depending on the interests of the local monitoring authority. The NRWQN design was intended to include upstream low-impact reference sites (i.e., natural land covers) as well as sites impacted by development (i.e., land use change), and to cover a large part of the country, but it is biased towards large streams and rivers. There tends to be low representation of reference sites and small streams overall, and sites often have mixed land use. Of relevance to this report, sites are spread around the catchments, including in upper and mid-catchments, rather than being focused on the outlet to the sea. Approximately 75% of the sites have flow monitoring sites nearby, which is useful for making corrections for the effect of flows in trend assessments.

Figure 13: Map of routine water quality sampling locations (blue points).

A range of water quality variables are measured, with some variation in the suite of variables between sites, depending on the authority doing the measurements. For example, nutrient concentrations and their speciation (such as total nitrogen, ammoniacal nitrogen, and nitrate) are measured at most sites, while heavy metals and indicators of faecal contamination (e.g., *E. coli*) are measured at only some sites). Monitoring of dissolved heavy metals (e.g., dissolved copper and dissolved zinc) is only routinely done at some sites within Auckland, Christchurch, Wellington, and the Taranaki region (Gadd et al. 2020b). The record duration also differs between sites. Over time, more standardisation of sampling, analysis, and reporting is progressing through the National Environmental Monitoring and Reporting

Standards (NEMS)[2], and more standardisation will occur as councils are required to report water quality compared with grading bands under the National Policy Statement for Freshwater Management (NPS-FM) (New Zealand Government 2020).

With regards to the measurement of suspended sediment at these national and regional sites there are three commonly measured variables i) total suspended solids (TSS) (or sometimes suspended sediment concentration (SSC)), ii) nephelometric turbidity, and iii) visual clarity. These three variables are all closely inter-correlated (Davies-Colley & Smith 2001, Davies-Colley et al. 2014, Hughes et al. 2014). Total suspended sediment is only measured at a subset of sites, and this is likely to continue to be the case because NPS-FM attributes are targeted at visual clarity of water and deposited sediment rather than sediment concentration. Despite it being a common measurement, nephelometric turbidity measurements of discrete grab samples are of very limited practical use. This is because turbidity is only a relative measure of light scattering (versus arbitrary standards) and has no intrinsic environmental relevance until it is calibrated to an actual scientific quantity (e.g., TSS or visual clarity) (Davies-Colley & Smith 2001). Furthermore, different sensors can output different turbidity values on the same suspension (Davies-Colley et al. 2021).

The regional water quality datasets are managed and archived by the regional councils/territorial authorities (and NIWA, in the case of the NRWQN) and are reported in various state of environment reports and online resources for each agency. Key data are collated nationally by LAWA (Land and Water Aotearoa)[3] and available through a web-based mapping interface and can be downloaded. Data are also collated by the Ministry for the Environment and Stats NZ for national state of environmental reporting. The most recent national reporting was in 2021 (Ministry for the Environment & Stats NZ (2022), while Larned et al. 2108b provides a detailed science synthesis of the data up till 2017.

In addition to the NIWA and regional council sources of sediment data mentioned above, another source of national scale sediment data from New Zealand catchments is the work carried out by Hicks et al. (2011) (and later updated by Hicks et al. 2019). For the purpose of calibrating their sediment load estimator model (described later in Section 4.1.2), Hicks et al. (2011) calculated mean annual suspended sediment loads for 233 rivers (Figure 14). For most rivers the loads were determined using the sediment rating approach (a detailed description and assessment of the sediment rating approach is presented by Horowitz 2003). Relationships were developed between field measurements of suspended sediment concentration (SSC) and discharge (i.e., river flow rate) obtained by multiple organisations (e.g., regional councils, NIWA, and power generation companies). These relationships were applied to the full flow record for each site to calculate mean annual suspended sediment loads. The SSC data were mostly collected using depth-integrating samplers using the methods described by Hicks & Fenwick (1994), while the river discharge data were obtained at the same time by current meter. As indicated in Figure 14, these sites are distributed throughout the country and many are large rivers that discharge directly to the coast.

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[2] www.nems.org.nz

[3] https://www.lawa.org.nz/

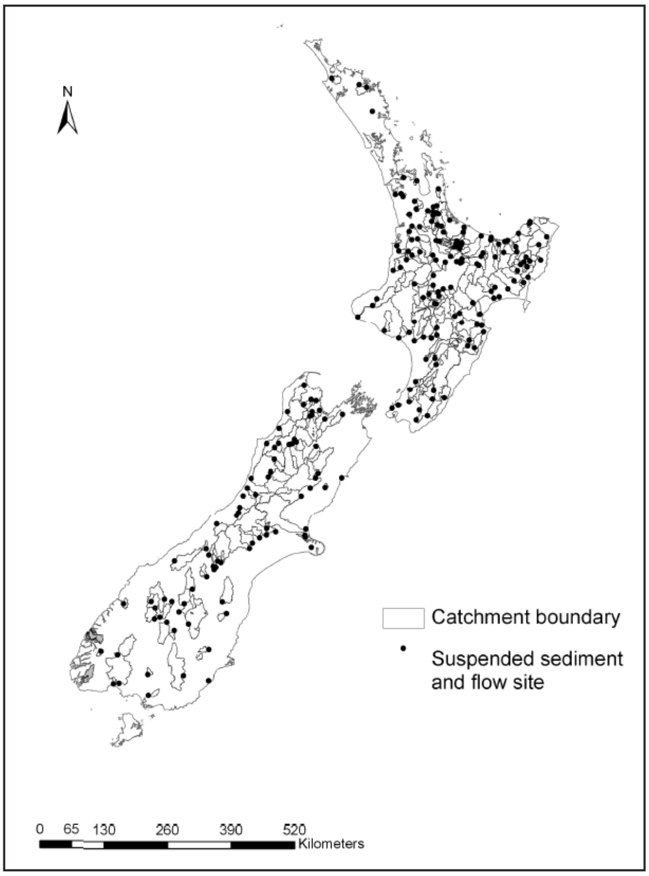

Figure 14: Locations of river monitoring sites where Hicks et al. (2011) estimated sediment loads using the rating curve approach. (Source: Hicks et al. 2011.)

Another national-scale source of stream water quality data is the Urban Runoff Quality Information System (URQIS). URQIS is a NIWA-compiled database that contains data on stormwater and urban stream water quality data. URQIS is a freely available web-based tool that can be queried by users[4]. URQIS allows users to obtain urban water quality information by land use, region, water type (treated or untreated stormwater, urban streams), and flow conditions (storm flow or baseflow) (Gadd et al. 2014). The database includes data supplied by city and regional councils, research institutes, transport agencies, and universities across New Zealand (Gadd et al. 2014). The database includes data on the following: solids (e.g., TSS/SSC), metals (dissolved and particulate forms), micro-organisms (e.g., *E. coli*), nutrients, hydrocarbons, PAHs, and miscellaneous measures of water quality (e.g., pH, dissolved oxygen, temperature) (Gadd et al. 2014).

Larned et al. (2018a) reviewed and summarised the rich literature on observations of land-use impacts on freshwater and estuarine environments in New Zealand. Most of the emphasis in the review was on the freshwater environments. The report put the information in a pressure-state-impact (PSI) framework, going beyond simply summarising data to investigate the role of pressures such as land use on state and relations between state and impacts such as macroinvertebrate community composition.

Among the various analyses and summaries, nutrient and sediment losses from paddock or small-catchment scale studies were summarised (Figure 15) showing larger nutrient losses from intensive pasture, with high nitrogen losses from dairy cattle, although lower sediment losses for dairy cattle

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[4] https://urqis.niwa.co.nz/#/report

(associated with lower land slope). The figure does not include market gardens, which can produce highest nitrogen yields (>100 kg ha^{-1}) (e.g., Francis et al. 2003).

Urban areas have TN losses in the order of 15 kg ha^{-1} (Moores et al. 2017), comparable with mixed pasture. Gadd et al. (2020a) reviewed the state and trends of water quality in New Zealand and found that monitored nitrogen concentrations in urban streams were comparable with those in pasture streams (typically a median concentration of 1 mg L^{-1} TN, predominantly in nitrate form, for urban streams).

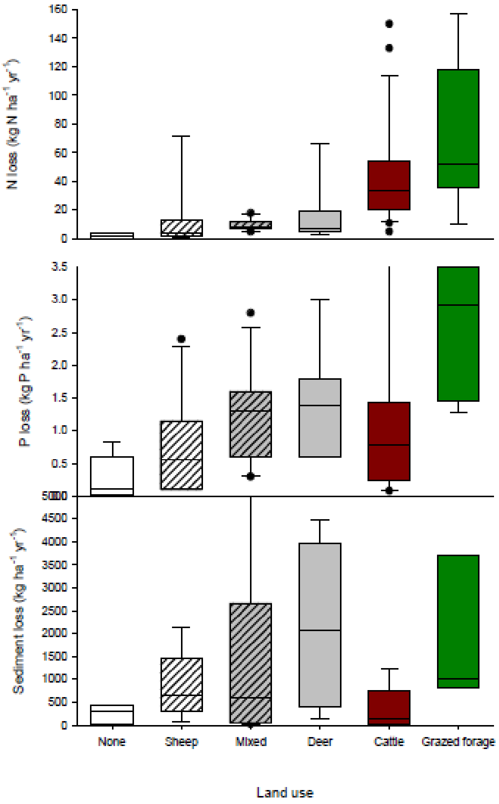

Figure 15: Summary of nutrient and sediment losses in small-scale observational studies in New Zealand. Box plots showing median concentration (central horizontal line). 25th and 75th percentiles (box), 10th and 90th percentiles (whiskers), and outliers (dots) for top) N, middle) P, and bottom) sediment annual loads for each pastoral stock class. 'None' refers to non-agricultural rural land uses, such as exotic plantation and native forest. 'Mixed' refers to catchments with more than one stock class. 'Cattle' refers to both dairy and beef stock classes. (Source: figure 3.2 of Larned et al. 2018a.)

Building on that report (Larned et al. 2020) published a paper on PSI focusing on fresh water, including consideration of land management practices. The authors noted that "*proportions of catchment area in urban and pastoral LULC were consistently, positively correlated with contaminant levels in water bodies and negatively correlated with ecological-health indicators… Correlations between proportions of catchment LULC classes and contaminant levels in rivers have been reported periodically for almost 30 years. The spatial extent of these correlations ranged from single catchments to multiple catchments distributed across New Zealand. The reported correlations were highly consistent: concentrations and loads of total N (TN), total P (TP), nitrate-N (NO3-N) and dissolved reactive P (DRP) were positively correlated with pastoral, urban and planted-forest land cover, and negatively correlated with natural (i.e. native forest, scrub, tussock) land cover.*" Limitations of the observational studies include: weakness of many of the relationships (despite their consistent direction across studies); predominant use of coarse land-use classes and categorical rather than continuous predictors; limitation to P-S, S-I, or P-I relationships; limitation to bivariate rather than more complex relationships; gaps in the range of variables and responses; and limitations in correlative rather than causal relationships for predicting future responses to land use or management.

4.1.2 Modelling of land-based stressors in New Zealand catchments

While there are numerous monitoring sites nationally, they do not provide a comprehensive view of contaminant loading to the coast because a) the monitoring sites are not at the coast and b) some catchments are not represented.

A number of catchment-scale models are now available for estimating mean annual loads and concentrations at river outlets to the coast. Such models are typically applied using the River Environment Classification (REC) network (Snelder & Biggs 2002). The REC network is a system for classifying river reaches based on physiographic factors (e.g., climate, topography, and land cover) that control spatial patterns in river ecosystems (Snelder & Biggs 2002). In version 2.5 of the REC network there are over 11 000 streams outlets to the coast, out of almost 600 000 river segments nationally, although nearly 7000 of those outlets are small headwater streams running directly to the coast. Model predictions are usually provided for each river segment.

The CLUES model (Elliott et al. 2016, Semadeni-Davies et al. 2020) has been used in catchments throughout New Zealand to predict loads and concentration of contaminants. More specifically, CLUES predicts mean annual loads of total nitrogen, total phosphorus, suspended sediment, and *E. coli*, and concentration of nutrients. CLUES estimates the sources from each REC sub-catchment nationally, and accumulates the sources downstream, accounting for attenuation. The sources are estimated from a simplified version of the farm-scale model Overseer (Grant et al. 2018) for pasture areas, and calibrated coefficients for other land uses, accounting for factors such as soil drainage, slope, and rainfall. The model is incorporated into software which is available for download from NIWA[5]. The underlying model structure is flexible. CLUES also calculates concentrations and summer loads (a time of the year when pollutants might have greatest downstream ecological effect), based on statistical relationships using the characteristics of the upstream catchment, and reports the relative contribution of different land uses to the total load. The model has been applied widely, for example, calculation of loads to the coast in response to land-use change (Parliamentary Commissioner for the Environment 2015), effect of mitigation measures such as stock exclusion (Elliott et al. 2020), estimation of historical loads, and the required load reduction to meet water quality targets. A recent example output for the Hauraki Gulf (Elliott et al. 2021) is show in Figure 16.

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[5] https://niwa.co.nz/freshwater-and-estuaries/our-services/catchment-modelling/clues-catchment-land-use-for-environmental-sustainability-model

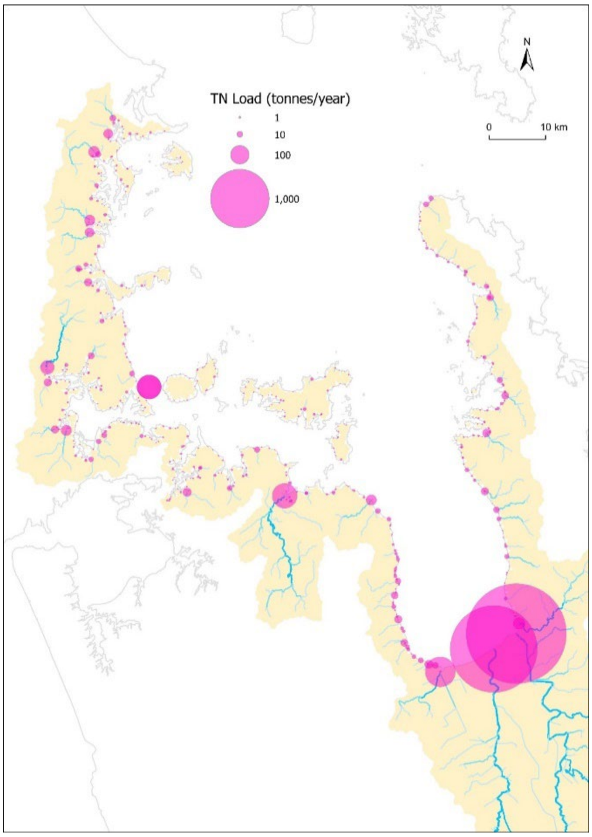

Figure 16: Mean annual Total Nitrogen loading from rivers to the Hauraki Gulf, from the CLUES catchment model and measurements. The load discharged from the Rosedale wastewater treatment plant on Auckland's North Shore is also shown.

CLUES also incorporates an estuary component, CLUES-estuary (Plew et al. 2020). Nutrient loads into the estuary are estimated from the basic CLUES model, which are then applied in an estuary mixing model to determine potential nutrient concentrations and then eutrophication response.

Snelder et al. (2018, 2020b) used statistical models (random forests, which accommodate nonlinearity and interactions) to relate mean annual nutrient yields (load in the stream divided by contributing catchment area) to characteristics of the upstream catchments, such as land cover, geology, climate, and, in the latest model, stocking intensity. The model was then applied across the country to predict loads for each REC river segment. Example applications include estimation of changes in load due to land development (Snelder et al. 2018) and required reductions in load to meet water quality targets. More recently, an estimate of stocking rate has been added as an explanatory variable. An example output, based on work discussed by Pearson et al. (2021) is shown in Figure 17.

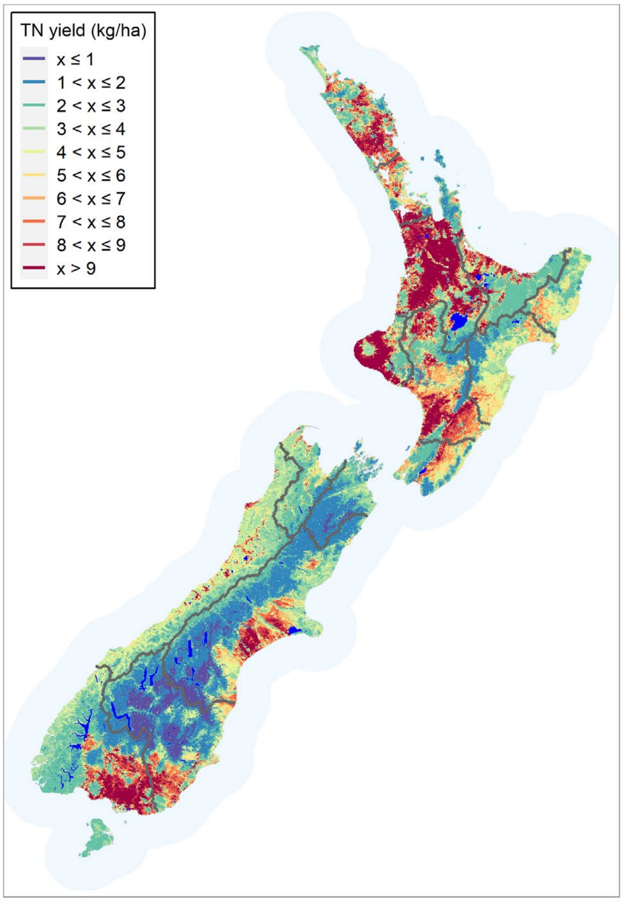

Figure 17: Nitrogen yields (load divided by upstream area) from statistical modelling. (Map provided by Ton Snelder, LWP Ltd., September 2021, pers. comm.)

Using simpler models relating loads to stream order (a measure of stream size) and REC river class, McDowell et al. (2017) determined that headwater streams made a large contribution to overall contaminant loads; a weakness of that work, however, is that the observed stream loads are only available for non-headwater streams.

Regional nutrient budget models are also being used by regional councils, such as in recent modelling for the Horizons Region (Snelder et al. 2020a) using the Contaminant Allocation and Simulation Model (CASM) and in modelling supporting the Waikato One Plan (Semadeni-Davies et al. 2015). The spreadsheet model uses source coefficients for land use in conjunction with catchment-lumped attenuation factors.

All the above models are based on mean annual loadings, with a seasonal correction possible in the case of CLUES. However, assessment of marine impacts could be impacted by the timing of delivery of loading to the coast. One approach to this assessment problem is to apply temporal disaggregation techniques based on concentration-flow and seasonality relationships in conjunction with predicted flows to break mean annual loads into daily loads, as done recently for the Hauraki catchment-coast modelling (Elliott et al. 2021). A difficulty with this approach is determining the concentration-flow relationship when there are no measurements. An alternative might be to use statistical estimates of the probability distribution of concentrations, but that also involves errors.

Instead of using mean annual budget-based models, dynamic simulation models are also being used in some regions to inform water quality management under the NPS-FM. An example is eWater Source[6] modelling being undertaken for the Greater Wellington Regional Council. Auckland Council have adopted the LSPC (Loading Simulation Programme C++) (Shen et al. 2005, Tech Tetra 2009) model for their Freshwater Management Tool, FWMT (Grant et al. 2018). NIWA, building on experience in small catchments, is using the SWAT model in the Hauraki catchment in a research project (Elliott et al. 2021). SWAT (Neitsch et al. 2005) has some advantages in that it simulates the soil-plant system, in comparison with some of the other models which, in the rural area, simply apply representative concentrations to the runoff (split into runoff components). However, such dynamic models have larger model input and calibration data requirements than budget models and require considerable expertise. They also are often not very reliable in their finer-scale time resolution predictions. A benefit is that there is usually more representation of contaminant generation and transport mechanisms, depending on the particular model. There is also the potential to apply such models over large areas. For example, the hydrodynamic model HYPE has been applied nationally to examine flow pathways, and SWAT has been set up nationally for sediment (Parshotam 2020). Data availability limitations are likely to be an impediment, though.

National-scale statistical models have been developed to predict nutrient concentrations across the REC, focusing on median and 95th percentile concentrations related to the NPS-FM, including species (different chemical forms) of nutrients. Similar to the statistical models of yield, these models represent the concentration as a function of catchment attributes, for national state of the environment modelling. The latest round of modelling for the Ministry for the Environment updated these models to 2021 (Ministry for the Environment & Stats NZ 2022). An example prediction of median concentrations is shown in Figure 18. As expected, concentrations are higher in areas with intensive pastoral vegetation and lower in mountain areas. One difficulty with such models is that of untangling the effects of co-varying predictors, such as climate and land use.

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[6] https://ewater.org.au/products/ewater-source/

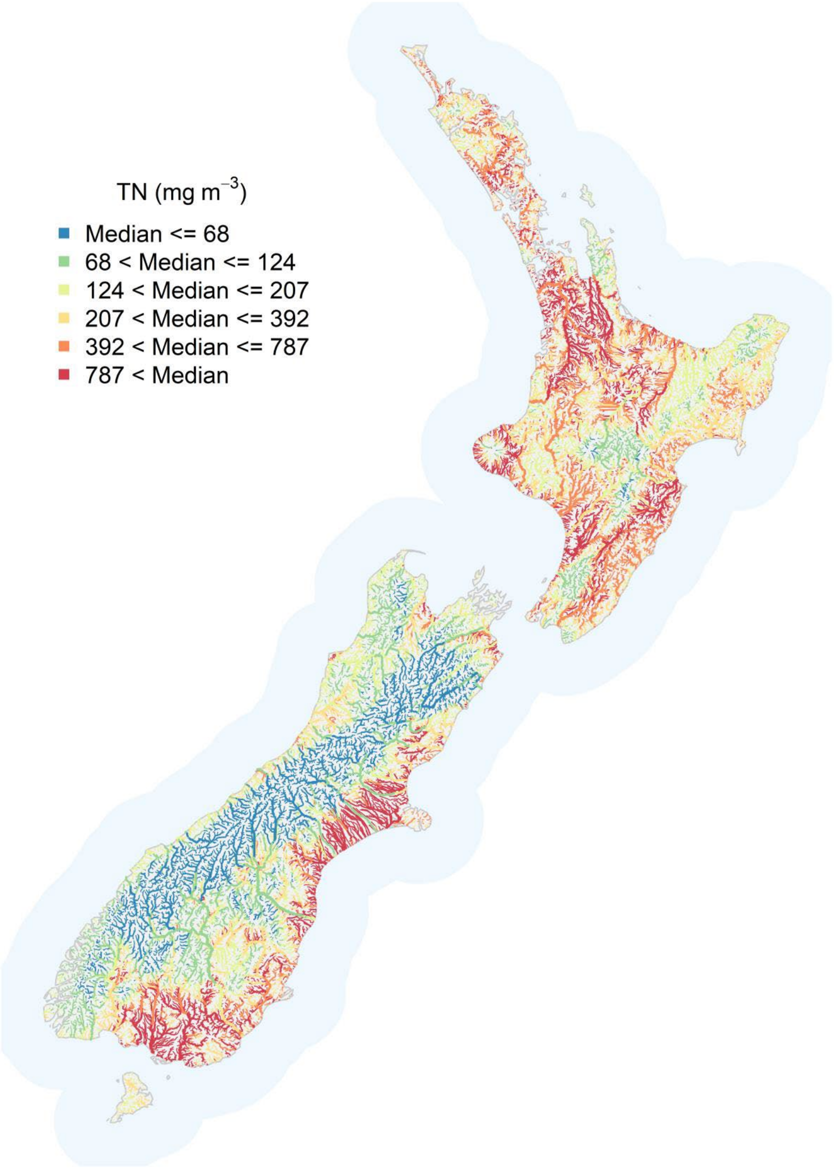


Figure 18: Predicted median total nitrogen concentrations from a national statistical model, from Whitehead (2018).

Seasonality in concentrations was modelled nationally by fitting models for each season (Whitehead et al. 2019). Example results are shown in Figure 19. Typically, nitrogen concentrations are slightly higher in winter than in summer, which could be for multiple reasons: lower N concentrations in deep groundwater that provides summer baseflow; greater activation of shallow flow pathways such as field drains in summer; and greater uptake into vegetation and increased denitrification over summer. Seasonality may have some relevance to marine receiving environment impacts because lower flux in summer coincides with periods of higher temperature and light conducive to eutrophication impacts, especially considering the lower flows in summer. Seasonality of flux was predicted by Elliott & Oehler (2009), showing similar effects overall.

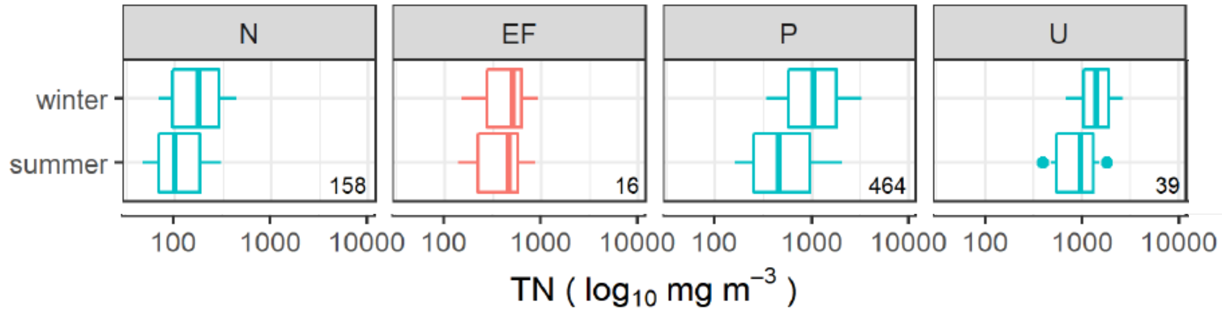


Figure 19: Seasonality of median total nitrogen (TN) concentrations at monitoring sites, by dominant land cover. N= native, EF = exotic forest, P = pasture, U = urban. The number of observation sites is shown as numbers inside each panel of the plot. (Source: Whitehead et al. 2019.)

Models have also been developed to predict reference conditions; that is, baseline conditions expected without land development (McDowell et al. 2013), nationally for each REC segment. The models fit measured concentrations as a function of pasture development, for each of selected REC river classes; the intercept for zero pasture development is the baseline concentration. Such methods give a gauge of the anthropogenic influence on water quality, and how low concentrations can reasonably be expected to go, given the climate, hydrological, and geological conditions. An example is shown in Figure 20. There is some variation between classes – for example, streams with cool-wet and cool-extremely wet catchments (CW and CX) have lower reference concentrations than streams with warm dry catchments in low areas (WDL), consistent reduced mineralisation and biological activity in cool areas, increased dilution in high-rainfall areas, and naturally-enriched soils in lowland depositional areas.

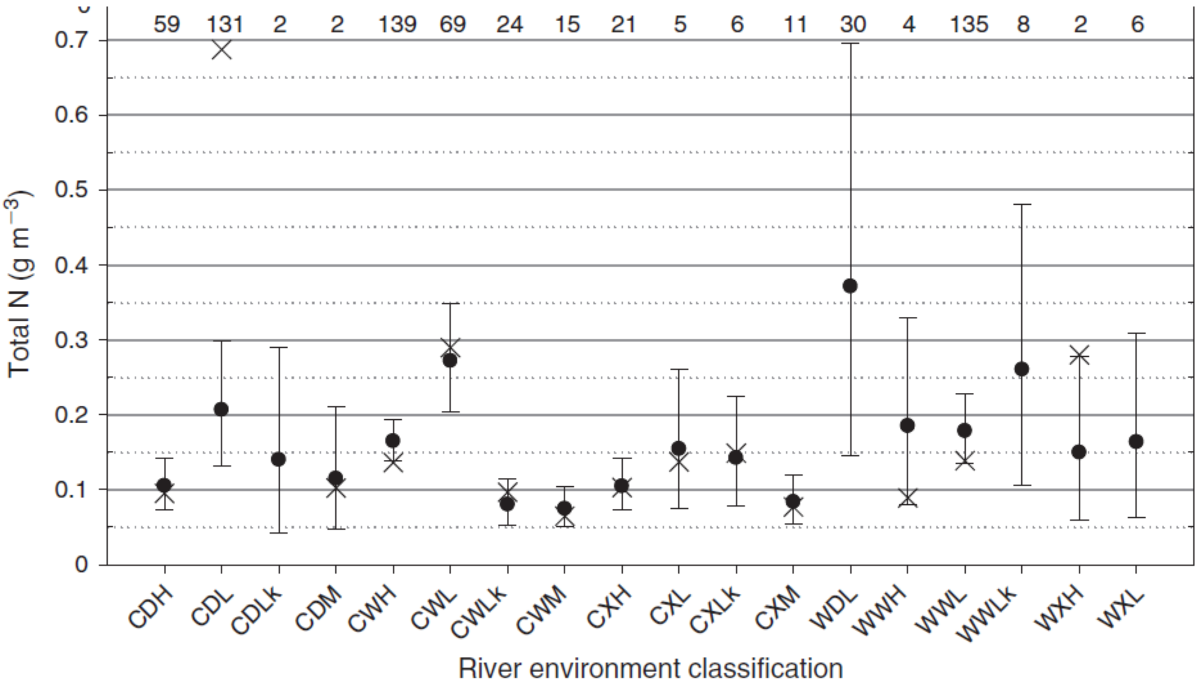


Figure 20: Estimated (circles, with confidence intervals) reference median total N concentrations grouped by REC (2nd level) class (from McDowell et al. 2013). Crosses show median measured concentrations with natural vegetation. Numbers are the number of monitoring sites use for regression in the class.

There are a number of models used in New Zealand catchments that are specific to the prediction of suspended sediment yields. Arguably, the most comprehensive attempt to assess river suspended sediment loads at the national scale from New Zealand rivers was carried out by Hicks et al. (2011). This work produced a national sediment yield GIS (Geographic Information System) grid from which the mean annual suspended sediment yield from any defined river catchment could be determined. The model relates suspended sediment yield from *driving* factors (related to the power generated by rainfall and runoff) and *supply* factors that relate to geological characteristics, erosion processes, soil type, and physiography (mainly slope) (Hicks et al. 2011, Hicks et al. 2019). The driving factors control the rate of erosion and the delivery of sediment through the landscape, while the supply factors control the availability of sediment in the landscape and the proportion delivered to river channels (Hicks et al. 2011, Hicks et al. 2019) (Figure 21). The model is empirically-based, therefore it does not specifically account for each of the individual processes (e.g., bank erosion, hillslope erosion, and mass wasting) that contribute sediment to river systems. The model was calibrated using sediment yield data determined from field-based measurements for the 230 catchments throughout the country (described in Section 4.1.1).

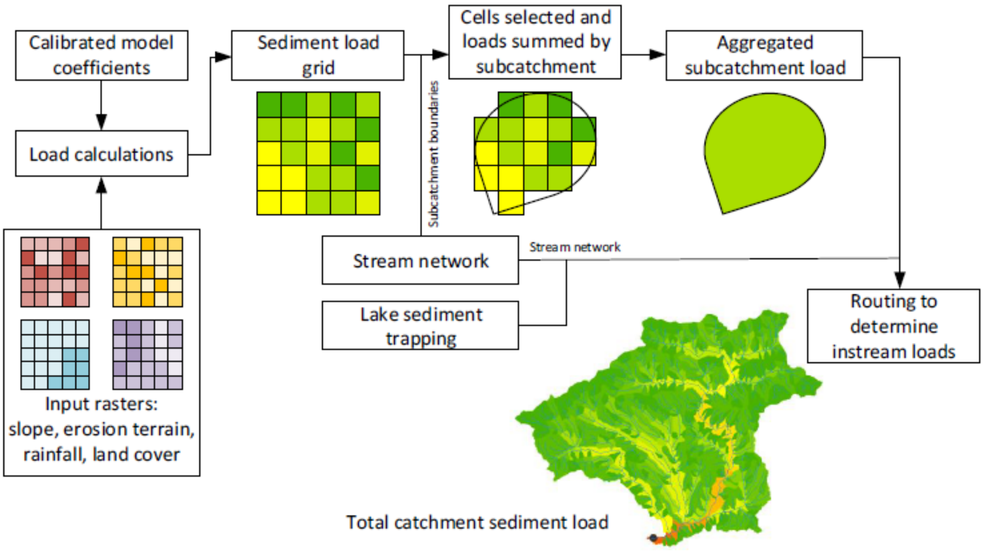


Figure 21: Steps of the 'Hicks' Sediment Load Estimator model. (Source: Hicks et al. 2019.)

Figure 22 illustrates the grid-based output of the sediment load estimator model, being the specific suspended sediment yield (total suspended sediment load divided by catchment area), as well as total loads delivered to the coast. Hicks et al. (2011) also presented the sediment yield (t y) for the main New Zealand rivers (Table 2). The total yield for all these rivers combined equalled 142.55 Mt y^{-1}. Hicks et al. (2011) argued that most of this sediment is discharged out to the ocean and little is trapped within estuaries. This is because the largest yield rivers (e.g., Waiapu, Waipaoa, Arawhata, Hokitika) have very small estuaries with very little opportunity for estuarine trapping.

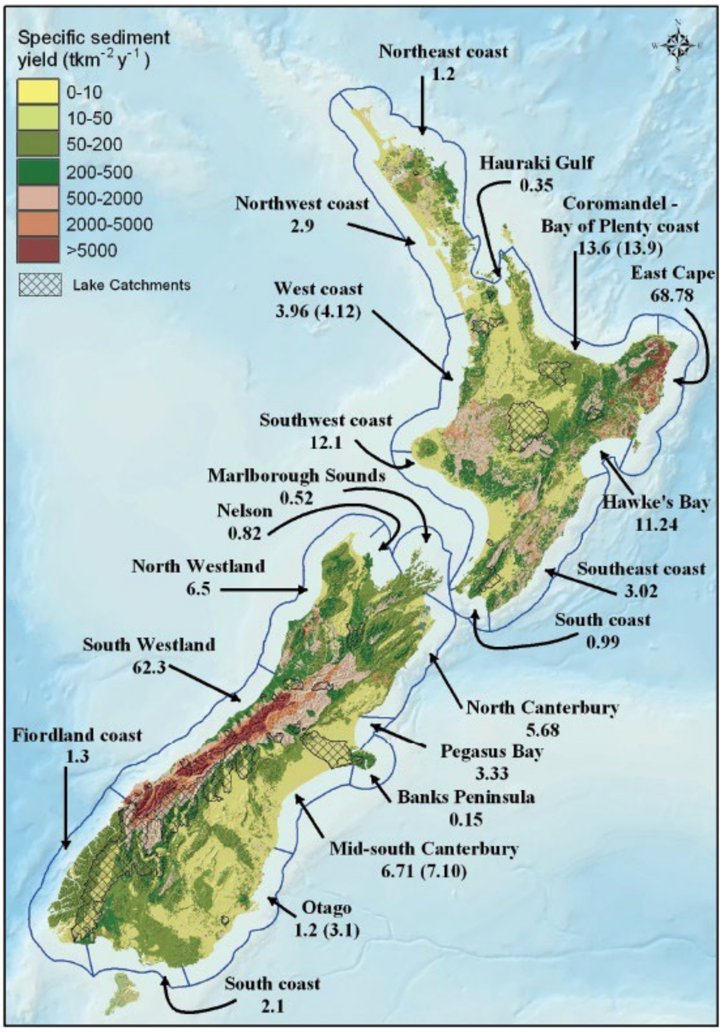
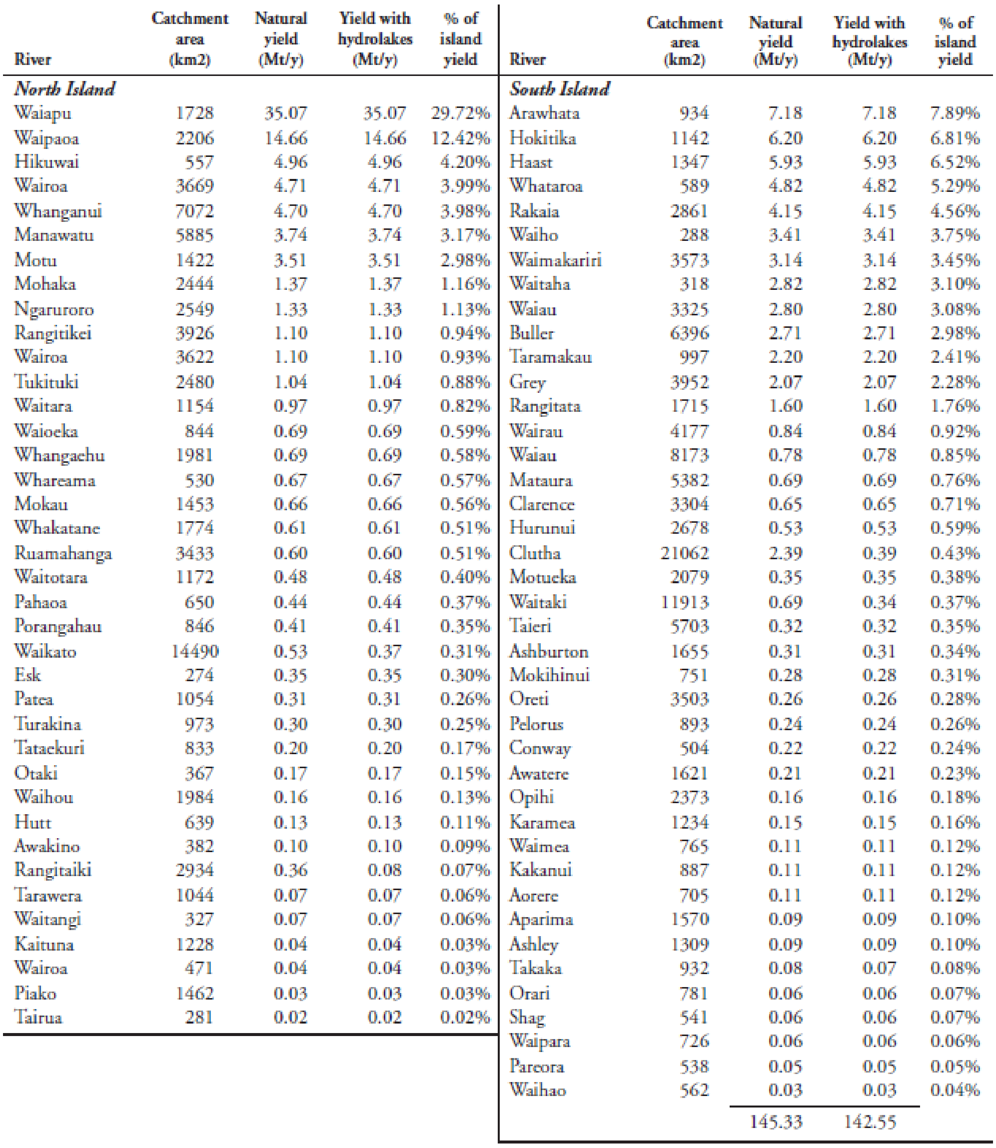


Figure 22: Map of specific suspended sediment yield as determined by the Hicks et al. (2011) sediment load estimator model. Sediment yield to the coast (Mt y^{-1}) for each region is indicated on the map. Yield totals are adjusted for sediment deposition lakes and reservoirs. Figures in brackets give yields without hydro reservoirs. (Source: Hicks et al. 2011.)

Table 2: Suspended sediment yields to the coast from some key New Zealand rivers. (Source: Hicks et al. 2011).



River	Catchment area (km2)	Natural yield (Mt/y)	Yield with hydrolakes (Mt/y)	% of island yield	River	Catchment area (km2)	Natural yield (Mt/y)	Yield with hydrolakes (Mt/y)	% of island yield
North Island					***South Island***				
Waiapu	1728	35.07	35.07	29.72%	Arawhata	934	7.18	7.18	7.89%
Waipaoa	2206	14.66	14.66	12.42%	Hokitika	1142	6.20	6.20	6.81%
Hikuwai	557	4.96	4.96	4.20%	Haast	1347	5.93	5.93	6.52%
Wairoa	3669	4.71	4.71	3.99%	Whataroa	589	4.82	4.82	5.29%
Whanganui	7072	4.70	4.70	3.98%	Rakaia	2861	4.15	4.15	4.56%
Manawatu	5885	3.74	3.74	3.17%	Waiho	288	3.41	3.41	3.75%
Motu	1422	3.51	3.51	2.98%	Waimakariri	3573	3.14	3.14	3.45%
Mohaka	2444	1.37	1.37	1.16%	Waitaha	318	2.82	2.82	3.10%
Ngaruroro	2549	1.33	1.33	1.13%	Waiau	3325	2.80	2.80	3.08%
Rangitikei	3926	1.10	1.10	0.94%	Buller	6396	2.71	2.71	2.98%
Wairoa	3622	1.10	1.10	0.93%	Taramakau	997	2.20	2.20	2.41%
Tukituki	2480	1.04	1.04	0.88%	Grey	3952	2.07	2.07	2.28%
Waitara	1154	0.97	0.97	0.82%	Rangitata	1715	1.60	1.60	1.76%
Waioeka	844	0.69	0.69	0.59%	Wairau	4177	0.84	0.84	0.92%
Whangaehu	1981	0.69	0.69	0.58%	Waiau	8173	0.78	0.78	0.85%
Whareama	530	0.67	0.67	0.57%	Mataura	5382	0.69	0.69	0.76%
Mokau	1453	0.66	0.66	0.56%	Clarence	3304	0.65	0.65	0.71%
Whakatane	1774	0.61	0.61	0.51%	Hurunui	2678	0.53	0.53	0.59%
Ruamahanga	3433	0.60	0.60	0.51%	Clutha	21062	2.39	0.39	0.43%
Waitotara	1172	0.48	0.48	0.40%	Motueka	2079	0.35	0.35	0.38%
Pahaoa	650	0.44	0.44	0.37%	Waitaki	11913	0.69	0.34	0.37%
Porangahau	846	0.41	0.41	0.35%	Taieri	5703	0.32	0.32	0.35%
Waikato	14490	0.53	0.37	0.31%	Ashburton	1655	0.31	0.31	0.34%
Esk	274	0.35	0.35	0.30%	Mokihinui	751	0.28	0.28	0.31%
Patea	1054	0.31	0.31	0.26%	Oreti	3503	0.26	0.26	0.28%
Turakina	973	0.30	0.30	0.25%	Pelorus	893	0.24	0.24	0.26%
Tataekuri	833	0.20	0.20	0.17%	Conway	504	0.22	0.22	0.24%
Otaki	367	0.17	0.17	0.15%	Awatere	1621	0.21	0.21	0.23%
Waihou	1984	0.16	0.16	0.13%	Opihi	2373	0.16	0.16	0.18%
Hutt	639	0.13	0.13	0.11%	Karamea	1234	0.15	0.15	0.16%
Awakino	382	0.10	0.10	0.09%	Waimea	765	0.11	0.11	0.12%
Rangitaiki	2934	0.36	0.08	0.07%	Kakanui	887	0.11	0.11	0.12%
Tarawera	1044	0.07	0.07	0.06%	Aorere	705	0.11	0.11	0.12%
Waitangi	327	0.07	0.07	0.06%	Aparima	1570	0.09	0.09	0.10%
Kaituna	1228	0.04	0.04	0.03%	Ashley	1309	0.09	0.09	0.10%
Wairoa	471	0.04	0.04	0.03%	Takaka	932	0.08	0.07	0.08%
Piako	1462	0.03	0.03	0.03%	Orari	781	0.06	0.06	0.07%
Tairua	281	0.02	0.02	0.02%	Shag	541	0.06	0.06	0.07%
					Waipara	726	0.06	0.06	0.06%
					Pareora	538	0.05	0.05	0.05%
					Waihao	562	0.03	0.03	0.04%
							145.33	142.55	

Another model that has the potential to provide detailed suspended sediment yield at a large scale is SedNetNZ, developed by Manaaki Whenua/Landcare Research specifically to estimate spatially-distributed catchment sediment loads. SedNetNZ is based on the original Australian model SedNet (Wilkinson et al. 2009), designed as a broad-scale model to provide relative approximations of catchment sediment yield and to identify general spatial patterns of erosion. It was originally developed for use at the regional scale (> 10^4 km^2) during the Australian National Land and Water Resources Audit (NLWRA; Prosser et al. 2001). It was subsequently applied at different scales throughout Australia (e.g., De Rose et al. 2003, Kinsey-Henderson et al. 2005). The New Zealand version of the model was adapted to include the key sources of catchment erosion in New Zealand catchments. SedNetNZ has been applied to several catchments including the Manawatu River (Dymond et al. 2016, Basher et al. 2020) and the Waipa River (Palmer et al. 2013). Two key advantages of SedNetNZ over some other catchment-scale models are: i) its ability to separately account for all erosion processes (e.g., bank erosion, hillslope erosion, and mass wasting), and ii) the ability to run scenarios to assess the effect of catchment mitigation measures. A disadvantage of the model is that detailed erosion source data (that may not readily available at the national scale) are required. Hence to date, it has not been applied at the national scale.

There are no national- or regional-scale urban water quality models in New Zealand since urban processes operate at the site to city-block scale. The CLUES model, which has a single urban land use class (and so provides only a very coarse representation of sediment and nutrient generation in urban areas) is not intended for urban contaminant modelling. Metals, which are key urban contaminants in New Zealand are not modelled by CLUES. The Auckland Council Contaminant Load Model (CLM; Auckland Regional Council 2010b, Auckland Regional Council 2010a) and the closely related NIWA Catchment Contaminant Load Model (C-CALM; Semadeni-Davies et al. 2010, Semadeni-Davies & Wadhwa 2014) include metals and have similar calculation methods to CLUES. Like CLUES, both models are steady-state models and are used to estimate mean annual contaminant loads (total and dissolved zinc and copper and sediments). This is done by multiplying the area of urban contaminant sources (i.e., roads, roofs, paving, grass, etc.) by yields that have been derived from measured water quality data (Timperley et al. 2005); these loads can be reduced in the models to represent different stormwater quality treatment options such as rain gardens, ponds, wetlands, and swales.

Most dynamic urban drainage models that include water quality modelling use some form of build-up /wash-off algorithm (Sartor & Boyd 1972, Sutherland & Jelen 2003); these include, for example, MIKE URBAN[7], SWMM[8], and Infoworks[9]. The amount of contaminant build-up at the start of a rainfall event is a function of the accumulation rate and period since the previous rainfall. The amount of wash-off during a rainfall event depends on the intensity, duration, and depth of the rainfall, the pH of the rain, the roughness and porosity of the road surface, and the physical properties (size, weight, shape) of the particles (Egodawatta et al. 2007, Egodawatta et al. 2013). These models tend to characterise urban areas by land use (e.g., commercial, residential, and industrial), although they can also be calibrated for urban surface types. For example, the Modelled Estimates of Discharges for Urban Stormwater Assessments (MEDUSA) model (Charters 2016, Charters et al. 2017) developed by the University of Canterbury uses a spatially distributed build-up/wash-off model to estimate loads discharged from urban surfaces, including roads and roofs, during rainfall. The model parameters were derived from water quality monitoring. Metal concentrations in runoff from urban surfaces, included in MEDUSA, have recently been investigated with an eye to updating the model (Charters et al. 2021). In MEDUSA, the total zinc and total copper loads are proportional to those determined from monitoring for TSS based on a strong correlation between the metals and TSS, and the parameters for dissolved zinc and copper are proportional to those for total zinc and copper, respectively. To date, the model is only available for Christchurch.

The Australian Model for Urban Stormwater Improvement Conceptualisation (MUSIC; eWater 2014) [10] uses a stochastic modelling approach rather than build-up/wash-off equations. Each source in the catchment is assigned a distribution of contaminant concentrations from which a concentration is drawn stochastically for each rainfall event, under the assumption that over a long time period, the concentration distribution seen in receiving waters will approximate the actual distribution of contaminant concentrations. The contaminant concentration distributions are assumed to be log-normal and are defined by the mean and standard deviation of event mean concentrations for rainfall events specific to the contaminant and land cover type. Sources are generally represented by land use, although it is also possible to model sources by land cover, particularly for roads. MUSIC has default concentration distributions for TSS, phosphorus, and nitrogen derived from Australian water quality data, but users can provide their own local concentration distributions for these contaminants or can swap TSS for other contaminants such as metals. Although this model has been used in New Zealand for urban stormwater planning, it has not been widely used for water quality modelling.

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[7] https://www.mikepoweredbydhi.com/products/mike-urban

[8] https://www.epa.gov/water-research/storm-water-management-model-swmm
https://innovyze.com/products/stormwater-sewer-flood-modeling/xpswmm/

[9] https://www.autodesk.com/products/infoworks-icm/

[10] https://ewater.org.au/products/music/

4.2 Catchment interventions aimed at mitigating land-based stressors

The implementation of mitigation measures to improve stream water quality and reduce the export of sediment and nutrients to downstream freshwater and marine receiving environments has become a popular approach over the last 40 years. Alternative terms for 'mitigation measures' are 'best management practices' (BMPs), 'good management practices' (GMPs), and 'land management interventions'. A wide range of mitigation measures are available to reduce land-derived pollutants and most of these have been applied to agricultural land (Table 3). Several summaries of these measures, their effectiveness, and cost are available (e.g., Hughes 2016, McKergow et al. 2016, McDowell et al. 2018, Melland et al. 2018). The mitigation approaches available for use within plantation forests, especially applicable during forest harvest activities, are comprehensively summarised by Bryant et al. (2007). Table 4 includes examples of the general hillslope erosion mitigation measures that are available for use during harvest activities.

Despite the popularity of water quality mitigation measures, in-field monitoring and reporting on their effectiveness is rare (Alexander & Allan 2007, Thompson et al. 2018). The dearth of such reporting probably reflects the necessary, but inconvenient, long duration of the mitigation effectiveness studies if they are to be meaningful. Often 'space for time' (SFT) type approaches are used to assess the impact of mitigation approaches. Instead of monitoring the impact of the implementation of a mitigation measure over a time, SFT studies compare differences between unimpacted sites and sites that have undergone some intervention a number of years prior. An example of such a study was that carried out by Parkyn et al. (2005). They assessed indicators of habitat quality and water quality at nine sites in North Island that had been fenced and planted (from 2 to 24 years prior) and compared them with unbuffered control reaches nearby. While SFT studies can be useful, they do have limitations and have been shown to perform poorly compared with studies that monitor the impact of mitigations over time (e.g., Before After Control Impact (BACI) studies; França et al. 2016).

Table 3: Example mitigation measures for sediment and nutrient loss from agricultural land.

Location	Mitigation measure examples
Farm management	Alter stock type and numbers (including wintering off-site) Modify pasture feed type (e.g., plantain, low-N grasses) Restrict grazing on feed crops Fertiliser amount, timing, type, and application method Dairy shed effluent management Animal housing Diet modification (to reduce N) Soil amendments (such as alum) to immobilise P Tile drain amendments Flood irrigation management (wash off from dyke irrigation) Nitrification inhibitors applied to land
Edge of field and stream network	Fencing to restrict stock access to streams Grass buffer strips Riparian planted zones Wood-chip bio-reactors Constructed wetlands In-stream P sorbents Detention bunds and dams Controlled drainage system

Table 4: Example sediment loss mitigation measures for available for use during forest harvest activities.

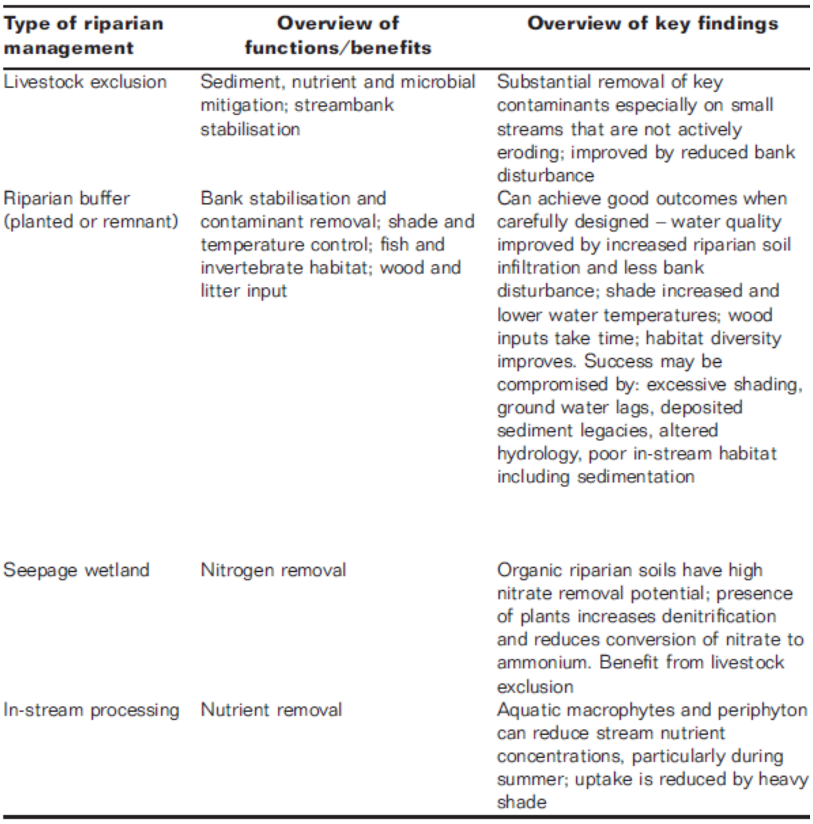
Erosion/sediment control mitigation approaches	Examples
Erosion control practices	Diversion channels and bunds Check dams Slope stabilisation Slash mulch application to bare ground Grassing and hydroseeding Surface roughening
Sediment control practices	Silt fences Straw-bale barriers Earth bunds Silt traps Sediment retention ponds

However, long-term field-based studies that assess the impact of the implementation of mitigation measures and compare this with pre-impacted results and sites that have not been impacted (e.g., BACI design studies) are globally rare. Within New Zealand there are two notable long-term studies: i) the Whatawhata Integrated Catchment Management Plan Project, and ii) the Dairy Best Practice Catchment Project. The Whatawhata Integrated Catchment Management (ICM) Project is the longest continuously monitored BACI catchment-scale study in New Zealand. The Whatawhata ICM Project used a BACI design to analyse the effects of land use changes on stream conditions. Land use changes implemented to improve stream water quality included planting of pine forests on steep unproductive land, exclusion of cattle from streams, riparian planting, poplar planting to control erosion, and reducing the number and size of cattle that are on hillslopes during winter. Multi-site monitoring (including nutrient and sediment) within three sub-catchments in the lower Waipa catchment in the Waikato Region commenced in 1995. The ICM-related land use changes occurred in the experimental catchment (Mangaotama catchment) in 2000–2001 and monitoring continued until 2020. There are numerous publications that report the findings from this study and some key recent ones include: Quinn et al. (2009), Hughes et al. (2012), Hughes & Quinn (2014), and Davies-Colley & Hughes (2020). These publications reported on the impact of various mitigation measures on concentrations and loads of sediment and various forms of nutrients, finding that while some measures of water quality improved, others remained static and some even appeared to degrade. An important conclusion from these studies is therefore that measures of stream ecosystem health and water quality do not always respond in a simple way to the implementation of sustainable land use interventions. The information collected during this study therefore provides valuable information that can be used to improve the performance of catchment-scale water quality models, which often make poorly informed assumptions about the response of catchments to management interventions.

The Dairy Best Practice Catchments (DBPC) Programme was initiated in 2001 by the New Zealand dairy industry to help improve understanding of the link between dairying land use and environmental outcomes and provide farmers with mitigation options (Scarsbrook 2011). The DBPC programme built upon a long-term monitoring and research site established at the Toenepi Stream in Waikato in 1995. With support from the Sustainable Farming Fund, regional councils, and central government, the programme expanded to four catchments in 2001 (Toenepi, Waikoura in Taranaki, Waikakahi in Canterbury, and Bog Burn in Southland), with a fifth (Inchbonnie on the West Coast) added in 2004. Mitigation measures implemented varied by catchment but included such approaches as riparian planting and fencing, farm effluent management, use of stand-off pads, and total exclusion of cattle from streams. The results from this programme have been reported by various publications, including: Monaghan et al. (2007), Wilcock et al. (2007), Wilcock et al. (2013), and Wright-Stow & Wilcock (2017). As with the long-term Whatawhata ICM study, the water quality response to the implementation of the mitigations measure was complex. Some improvements in water clarity and reductions in TSS concentrations were reported; however, no improvements in nutrient concentrations were observed (Wright-Stow & Wilcock 2017).

Arguably, one of the most widely applied approaches for the reduction of land-derived contaminants (including sediment, nutrient, and micro-organisms associated with faecal matter) is the application of riparian management within catchments. This can be attributed to the fact that improved riparian management practices, such as the exclusion of stock from waterways and the planting of riparian areas, can (depending on location and physiographic factors) provide a number of benefits including enhanced nutrient processing, stream bank protection from erosion, shading small streams for temperature control, providing leaf and wood input to stream ecosystems, and enhancing fish and invertebrate habitat. A recent review by McKergow et al. (2016) described in detail the use of riparian management approaches in New Zealand catchments. They documented the history of riparian management in New Zealand and summarised the evidence indicating its effectiveness (Table 5).

Table 5: Findings from New Zealand studies of riparian management. (Source: McKergow et al. 2016.)



Type of riparian management	Overview of functions/benefits	Overview of key findings
Livestock exclusion	Sediment, nutrient and microbial mitigation; streambank stabilisation	Substantial removal of key contaminants especially on small streams that are not actively eroding; improved by reduced bank disturbance
Riparian buffer (planted or remnant)	Bank stabilisation and contaminant removal; shade and temperature control; fish and invertebrate habitat; wood and litter input	Can achieve good outcomes when carefully designed – water quality improved by increased riparian soil infiltration and less bank disturbance; shade increased and lower water temperatures; wood inputs take time; habitat diversity improves. Success may be compromised by: excessive shading, ground water lags, deposited sediment legacies, altered hydrology, poor in-stream habitat including sedimentation
Seepage wetland	Nitrogen removal	Organic riparian soils have high nitrate removal potential; presence of plants increases denitrification and reduces conversion of nitrate to ammonium. Benefit from livestock exclusion
In-stream processing	Nutrient removal	Aquatic macrophytes and periphyton can reduce stream nutrient concentrations, particularly during summer; uptake is reduced by heavy shade

Another recent study by Hughes (2016) examined the effectiveness of stock exclusion and riparian planting at reducing stream bank erosion (hence sediment delivery to streams). This study found that there are very few studies that have used quantitative techniques to determine the impact of riparian management on stream bank erosion. Most studies reported that managed stream banks were in better condition than unmanaged banks and that the exclusion of livestock from riparian areas was the principal factor in the measured improvements or differences. Only two studies specifically attributed reduced stream bank erosion to the presence of riparian vegetation. Hughes (2016) concluded that more quantitative studies are required to adequately determine the effectiveness of riparian management interventions for reducing bank erosion.

There are a number of experimental edge-of-field type approaches that have shown potential to remove nutrients and sediment from both urban and agricultural catchments. Examples of such approaches include constructed wetlands and sediment detention ponds. Constructed wetlands attempt to simulate the pollutant attenuation ability of natural wetlands (Uuemaa et al. 2018). Constructed wetlands have been researched at several New Zealand sites. Studies from both New Zealand and overseas have demonstrated that well-designed wetlands are capable of significantly reducing nitrate and contaminant concentrations (e.g., Tanner et al. 2005, Tanner & Sukias 2011). Depending on location, constructed wetlands are capable of removing between 25 and 50% of the total nitrate delivered to it (with larger wetlands relative to their catchment area being more efficient) (NIWA 2021). Constructed wetlands have been shown to be less effective at attenuating phosphorus.

Sediment detention ponds have been used in the urban environment for many years and have been shown to be effective at reducing sediment loads, although dissolved materials are less effectively intercepted by them (Stanley 1996, Pettersson 1998). Recently, in New Zealand the idea of using ponds to remove land derived pollutants has recently been extended to rural areas in the form of detainment bunds (Levine et al. 2021a, Levine et al. 2021b). Detainment bunds are stormwater retention structures constructed of consolidated soil at the bottom of a hillslope with convergent flow paths. Detainment bunds intercept runoff generated during intense rainfall events and retain the runoff in a temporary pond (usually for a few days). While in the pond, particulate matter and the associated nutrients settle out and are retained in the pasture while the water drains away. A proof of concept study was carried out at two sites with the Rotorua district and it was demonstrated that the detainment bunds were able to intercept up to 60% of the sediment delivered from their catchments (Levine et al. 2021a).

There are a number of erosion and sediment control methods available that are specific to forestry activities (Bryant et al. 2007). Arguably, the forestry activities with the greatest potential for increasing sediment delivery to streams are: the establishment of forestry roading, the construction of landing sites, haul tracks, and firebreaks, preparation of land for forestry, and harvesting activities. Bryant et al. (2007) explains approaches for mitigating the impacts these activities, such as the avoidance of large bare earth areas for long periods of times, avoiding forestry activities near waterways, the management of forestry slash and residue, and the management and maintenance of hillslopes in the post-harvest period. The implementation of these practices is likely to reduce impacts of forestry activities. However, as with mitigation measures applied to other agricultural activities, studies that have actually quantified the impacts of these mitigation measures are rare. This is a situation which may improve over the coming years. The Ministry for Primary Industries (MPI) has partnered with a forestry company (OneFortyOne New Zealand) to carry out a study that assesses the performance of control practices aimed at reducing erosion and sediment delivery to waterways from forest harvesting (MPI 2021).

5. MECHANISMS OF ECOLOGICAL 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 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.

5.1 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.

5.1.1 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: Lloyd 1987, Newcombe & MacDonald 1991, Bash et al. 2001, Berry et al. 2003, Gibbs & Hewitt 2004, Thrush et al. 2004) 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, with work in northern New Zealand estuaries showing amphipods are an important juvenile fish diet component (Lowe et al. 2015), as well as for juvenile blue cod in the Marlborough Sounds region (Chang 2021).

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).

Several species of shellfish have been found to 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 1200 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 an 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 Norkko et al. (2002), Lohrer et al. (2004), and 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, 2006a). 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 via new recruitment and immigration in of adults. 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 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 is based on freshwater (in particular salmonid) species (see reviews: Moore 1977, Bruton 1985, Lloyd 1987, Newcombe & MacDonald 1991, Kerr 1995, Newcombe & Jensen 1996, 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, Confer et al. 1978, Gardner 1981, Berg & Northcote 1985, Barrett et al. 1992), or by reducing fish feeding rates (Sigler et al. 1984, Berg & Northcote 1985, Redding et al. 1987, Gregory 1993). The feeding of juvenile coho salmon (*Oncorhynchus kisutch*) dropped by 45% at a turbidity of 100 NTU (Reid 1998), and reduced growth rates associated with increased suspended sediments have been documented for Arctic 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 in 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 1982, Sigler et al. 1984, Berg & Northcote 1985, Lloyd et al. 1987). McLeay et al. (1987) found that Arctic grayling were displaced throughout a six-week experimental period at suspended sediment concentrations of 300 mg l^{-1} or greater, and 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 turbidity (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 by 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 turbidity levels 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 fish’s search volume, foraging strategy, and the influence of turbidity on prey behaviour to detect and avoid predators (Macia et al. 2003, Meager et al. 2005). 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 nonvisual ‘encounter rate feeding’ mode for turbid conditions (Marais 1984, Hecht & Van der Lingen 1992).

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 (Kerr 1995, Bergstedt & Bergersen 1997). At very high turbidity levels, 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), and 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 colouration (McLeay et al. 1987), 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% because of exposures to 500 and 1,500 mg l^{-1} respectively (Servizi & Martens 1987). Decreased tolerance rates to disease and time to death because of other environmental stressors have also been recorded for elevated suspended sediment concentrations (McLeay et al. 1984, Redding et al. 1987, Appleby & Scarratt 1989).

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 utilisation, 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 1240 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 1500 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'Connor et al. 1977). Larger particles were trapped by gill lamellae and stopped the passage of water leading to asphyxiation (Sherk et al. 1974). Another 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 were generated for six estuarine species by Sherk et al. (1974, 1975) using fuller's earth, and 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 1000 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.

5.1.2 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 worldwide global extent of 500 000 km^2. Numerous studies and accounts from many countries and regions consistently identify a long-term, worldwide 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. Lotze 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 6726 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 1996, Davison & Hughes 1998, Hemminga & Duarte 2000, Duarte 2002, 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 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 (Kautsky et al. 1986, Lumb 1989,). Sedimentation 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 of water quality on global kelp forests, there are also few 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 1996).

5.2 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 mat 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 invertebrate in-fauna 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 gobys' 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 prey 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.

5.2.1 Varied ecosystem response to eutrophication

Cloern (2001) reviewed the concept and consequences of coastal eutrophication and suggested that different attributes influenced how ecosystems responded to coastal eutrophication. Four 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) than those 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 (Meeuwig 1999).

5.2.2 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. 1990). 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, Levin 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 watershed. 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 had doubled, the intensity of *S. calicola* blooms was greatly reduced, and the seagrass *Halodule wrightsii* began to colonise parts of the bay (Johansson & Lewis 1992).

5.2.3 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 papilloma's 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.

5.2.4 Eutrophication in New Zealand

There is little (if any) work in New Zealand on how eutrophication impacts on coastal fisheries. The closest is the work of Savage (2009), 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 (filter-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. 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 showed 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 means that the principal role of freshwater entering the Nelson bays may be driving their estuarine circulation, density stratification, and turbidity, with their associated influences on light and nutrient availability for primary producers.

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.

5.3 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). Fresh water 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 the salinity of brackish water to fresh water 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 et al. 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 they 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, whereas 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 five-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%), whereas 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), whereas 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, and stable isotope analyses showed 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.

5.3.1 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 Section 6.2.4).

5.4 Effects of ground-water discharge

At local scales, submarine groundwater discharge (SGD) can approach or even exceed 50% of total freshwater inputs to coastal bays (Valiela et al. 1990, Slomp & van Cappellen 2004). This SGD fresh water can hold high concentrations of nutrients. For instance, work in a temperate bay by Sugimoto et al. (2015) found that 65% of the total dissolved inorganic phosphorus delivered by fresh water to the bay came from SGD. More broadly, positive correlations have been shown between high levels of SGD and elevated primary production in coastal waters in different coastal waters around the world (Miller & Ullman 2004, Hwang et al. 2005, Sanders et al. 2011, Sugimoto et al. 2015). The influence of SGD on higher trophic levels, including fisheries production, is only now beginning to be examined.

Hata et al. (2016) looked at the occurrence of juvenile marbled sole (*Pseudopleuronectes yokohamae*) and its prey species around submarine groundwater seepage on a tidal flat in southwestern Japan. This sole species occurred in coastal waters in the western North Pacific and is highly dependent on shallow water nursery areas. Larvae settle at about 10 mm total length in shallow waters (< 10 m) and prey on crustaceans and polychaetes for several months, until leaving for deeper waters at around 80–100 mm in size (NB: this is similar to 0+ snapper in New Zealand, although their depth distribution can extend to 30 m depth). They show high tolerance to low salinity conditions in laboratory experiments (Wada et al. 2007, 2011) and aggregation around an area with SGD reported (Ito & Nakagawa 2002). Hata et al. (2016) used radon-222 (^{222}Rn) as a marker for the presence of groundwater seepage. Radon-222 is a naturally occurring radioactive gas and occurs in concentrations 2–3 orders of magnitude higher in groundwater than surface waters (Church 1996, Kim et al. 2005). Its half-life is about 3.8 days (Taniguchi et al. 2002, Swarzenski et al. 2007, Charette et al. 2008, Dimova et al. 2009, Santos et al. 2010).

Surveying of gas concentrations on a subtidal flat was used to select sites across a radon-222 gradient, and juvenile sole and their prey taxa were surveyed from stations along the gradient. Offshore stations had both the highest gas concentrations, and the lowest salinities, as well as the highest chlorophyll-*a* levels (a measure of primary productivity). Marbled sole density was highest at the station where high submarine groundwater seepage was indicated, and the densities of prey taxa was also highest in this area. A negative correlation between gammarids and salinity was also seen (but not for the other major prey taxa). Stable isotope analysis found that post-settlement sole were more dependent on small crustaceans, with low δ13C values indicating that terrestrially derived nutrients were contributing to sole production.

Fujita et al. (2019) also used juvenile marbled sole as a species to provide direct experimental and survey evidence that fish feeding and growth are elevated by submarine groundwater discharge (SGD). Three shallow water sites around Japan were investigated, in the northwest Pacific, Pacific Ocean, and Seto Island sea regions (spanning > 1000 km of coastline). At each site, benthic microalgae production and gammarid crustacean abundances were compared between two stations with different levels of ^{222}Rn concentration and water salinity (as indicators of SGD), as well as surveys of small fish species. At one of the sites, a caging experiment was deployed, holding cultured juvenile sole for a 2-week period (16 cages per site, 1 sole per cage), across one station with high SGD indicator values, and another with low SGD indicator values. Mooring time series surveys were deployed at these two stations over a 24-hour period to measure depth, temperature, salinity, ^{222}Rn, SGD rate (using heat sensors), nutrients (NO^{-3}, NO^{-2}, NH^{+4}, $PO3^{-4}$), DIN (the sum of NO^{-3}, NO^{-2}, and NH^{+4}), DIP ($PO3^{-4}$), and chlorophyll-*a* (Chl-*a*).

At all three sites, benthic microalgae production was significantly higher at the stations with higher SGD signals. Gammarid amphipod and fish densities were also significantly higher at higher SGD signal stations. Each site was dominated by a different fish species, being Japanese sillago (*Sillago japonica*), surfperches (*Embiotocidae* spp.), and red seabream (*Pagrus major*) (Note, this seabream species is very similar to New Zealand's snapper *Chrysophrys auratus*). For the two stations used in the caging experiment, the station selected as having higher SDG indictors had significantly higher 24-hour mooring monitoring data for mean hourly ^{222}Rn concentrations, mean hourly SGD rate, salinity, DIN, and DIP; but not for temperature or chlorophyll-*a* concentrations. For the caged sole, stomach content weights were significantly higher at the higher SDG station, as was fish growth rate (mm per day). The mean stable carbon isotope ratio of juvenile marbled sole in cages at the end of the experiment from the higher SGD indicator site was intermediate between the values of the cultured sole still in captivity (i.e., cultured fish not used in the experiment) and the values of wild-caught fish from the site; indicating a shift in the caged fishes signature towards that of the site. There was also a significant difference in the stable carbon isotope of wild-caught sole between the two caging stations. This supported the conclusion that nutrients of terrestrial origin provided via SGD were utilised by juvenile marbled sole, through the benthic food web (especially gammarids).

Similar work was done on juvenile gray demoiselle *Chrysiptera glauca* at Mauritius (off East Africa), contrasting two rock pools with one being characterised by visible freshwater SGD, and the other being strictly marine in salinity (Lilkenday et al. 2019). This species is largely herbivorous, with some animal material also being consumed. Monthly monitoring of the two pools found temperature and pH were both higher in the 'SDG' pool, as were nitrite, nitrate, and silicate concentrations. Salinity was markedly lower in the SDG pool, while both pools were generally supersaturated with oxygen. Dissolved oxygen content was higher in the non-SDG pool, as were chlorophyll-*a* levels. The SDG pool was devoid of sessile algae, whereas the marine pool was largely covered by macroalgae and turf algae. Demoiselle densities between the two pools were not significantly different. Almost all juveniles examined (for both pools) fed on a mix of filamentous green algae, crustacean copepods, molluscs, and plant detritus. There was no difference in a fish condition index (weight-at-length) between the two pools. Growth models based on daily growth ring analysis showed that somatic growth rates were significantly higher for juveniles from the SGD influenced tide pool, versus the marine pool (Figure 23). Overall, it was concluded that physiologically beneficial environmental conditions brought about by the submarine influx of cold acidic freshwater enabled juvenile fish to exhibit elevated growth rates in the SGD influenced tide pool (Lilkenday et al. 2019).

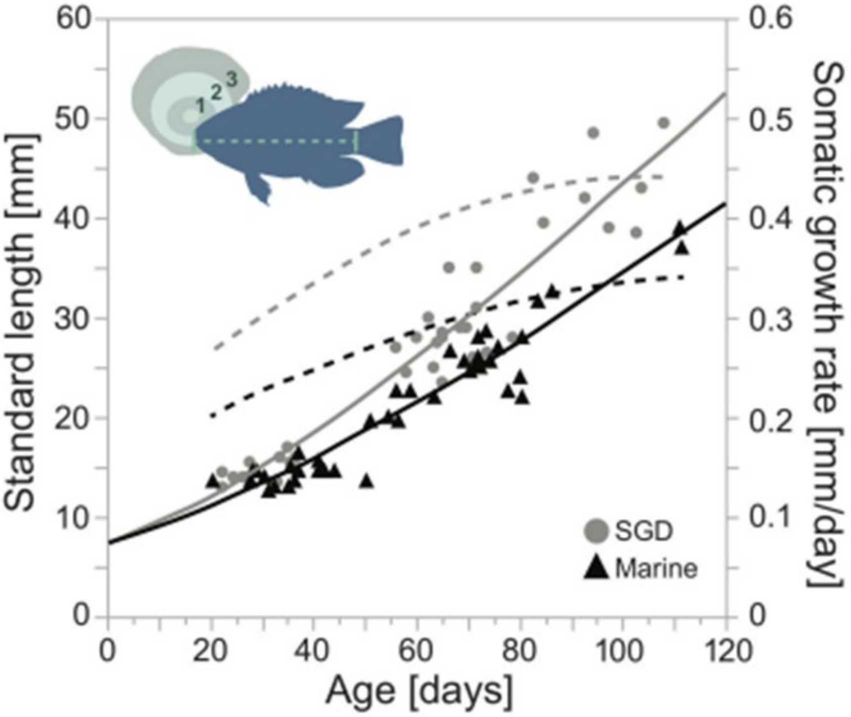


Figure 23: Juvenile gray demoiselle *Chrysiptera glauca* length-at-age and somatic growth rates, contrasted between a SGD dominated tidal pool (circles) and a marine water dominated tidal pool (triangle). (Source: figure 4 of Lilkenday et al. 2019.)

Pisternick et al. (2020) investigated the effects of SGD at the fish assemblage level, within a lagoon system of the island of Mauritius. This small tropical volcanic island (1860 km^2) has a relatively large human population of both residents and tourists. Sugar cane plantations are also a dominant industry, covering 40% of the island. The combination of sugar production, increasing tourist numbers, poorly developed wastewater management, and sewer systems are thought to be the major anthropogenic nutrient sources to its coastal waters (Ramessur 2002). On the north-western coast, six distinct groundwater springs discharge into the southern part of the island's lagoon system. These groundwater springs are connected to the 'Northern Basin aquifer', which covers around 200 km^2. Three sites were investigated inside the lagoon: one encompassing the six distinct groundwater springs, one highly influenced by freshwater influx through the springs, and one strictly marine control site. Using remote underwater video surveys, the groundwater springs site was found to support significantly higher fish abundances than the other two sites. Both the springs/springs-influenced sites were characterised by elevated water nutrient loadings, and higher macroalgae cover; whereas the marine site was characterised by higher water salinity and pH levels. It was concluded that there was evidence for "*a fresh SGD-driven relationship between altered hydrography and distinct fish communities with elevated abundances at groundwater springs in a coral reef lagoon*". Pisternick et al. (2020) recommended that the "*management and assessment of secondary consumer productivity in tropical lagoons should consider the effects of groundwater springs*".

Similar work in Tahiti's lagoon also found higher fish abundances around submarine springs, although fish were not identified to species. Sampling of zooplankton (dominated by copepods) also suggested (slightly) higher abundances around the submarine springs, although the difference was not statistically significant (Starke et al. 2020).

Burnett et al. (2019) evaluated the trade-offs between economic gains from using SGD to melt snow in Obama City, Japan; versus predicted negative impacts on nearshore fisheries productivity (Figure 24). In 2011, 13% of total groundwater use was for melting snow during the winter. The mixture of cold and warm currents in Obama Bay, combined with SGD nutrient supply, supports a very productive nearshore fishing industry. It was so productive historically that it was recognised as "miketsukuni"; a region responsible for supplying food to the imperial Japanese court. Using both hydrological and broad ecological/fisheries models, and fish prices, they estimated a 50% increase in groundwater pumping for melting snow would: a) decrease total SGD (1 million m^3) to the bay by 5% (50 000 m^3 per year); and b) decrease fish catch by 0.48 tons year, worth 457,393 JPY. If variable operating and capital costs of fishing were deducted, the annual net loss would be 189,003 JPY ($NZ2,356, at 0.0124 exchange rate). Allowing for these costs, the annual net benefit of using 50% more SGD for melting snow (relative to other more costly melting alternatives) ranged from 10.9 to 547.7 million JPY/year (NZ $135K–$6.8M).

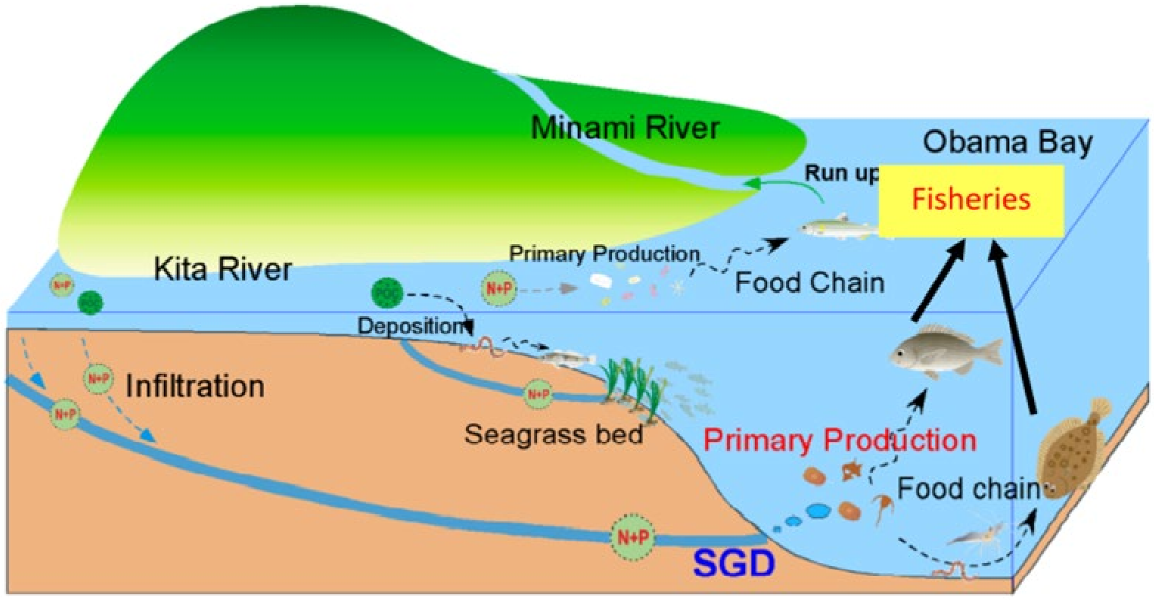


Figure 24: Link between submarine groundwater discharge (SGD) and fisheries productivity in Obama Bay. Note the potential link drawn to seagrass meadows. (Source: figure 3 of Burnett et al. 2018.)

5.5 International systems-scale work on land-based impacts

Holt et al. (2012) investigated the relationship between terrestrial landscape change (specifically the development of tree plantations), patterns of chlorophyll-*a* in nearshore coastal waters, and the biological condition of the commercially important shellfish *Concholepas concholepas* (known as *loco*) in southern Chile, South America. While called the Chilean abalone, it is not a true abalone but a member of the family Muricidae, also known as murex snails or rock snails (Figure 25). It is carnivorous, with mussels being a dominant prey.

Figure 25: The Chilean abalone *Concholepas concholepas* (*loco*).

Locos are economically the most important shellfish for artisanal fishers in Chile. Following a collapse of the *loco* fishery in 1990, a new management system was set up in which fishers were assigned exclusive fishing rights to benthic resources in set local management areas. Each area is surveyed is run through a local syndicate, and each year a baseline study of each management area (carried out by nationally certified consultants) is run to provide information on *loco* populations (including population size, size structure, and biomass) (Bitecma 2002). Harvest quotas are suggested for each local area and vetted by a national fisheries agency. Fishers cannot harvest *locos* within an area without this ecological evaluation, and fishers must pay consultants to assess the *loco* population annually, although the government may subsidise this activity. Fishers use hookahs (divers using long air hoses from surface air compressors) in teams of two to four to harvest *locos*. For the region used in this study, artisanal fishers harvest an annual average of 3278 tons of *locos* (2003–2005 average) with 23 000 artisanal fishers participating.

Locos (n=1374) were sampled across 13 watersheds (35 853 km^2) and 42 fisheries management areas (spanning 250 km of coastline). These shellfish were quantified for the weight and health of the *loco*, as well as for the fouling organisms associated with their shells. Landsat satellite imagery was analysed to create land-use classification, with a key focus on forests (planted and natural). Chlorophyll-*a* concentrations were extracted from SeaWiFS satellite images (9 km resolution). *Locos* harvested from management areas influenced by tree plantations had approximately 30% more endobiont (shell-boring) phoronids (horseshoe worms), almost twice as many endobiont polychaete worms, and twice as many epibiont (shell-attaching) barnacles than *locos* from areas in close proximity to watersheds dominated by native forests (15–20% of the watershed). Phytoplankton levels in the tree plantation influenced areas were higher, and eutrophication was inferred to be positively influencing the shell-dwelling filter-feeding epifauna.

Phoronid infested *locos* from coastal waters adjacent to watersheds with tree plantations were of relatively poor biological condition (smaller and narrower in width) and of reduced market value. The price of *locos* is based primarily on *loco* length and weight, so *locos* that weigh less than predicted based on a given length (*locos flacos*) result in lower prices paid to fishers. Higher polychaete and phoronid cover also reduces prices (Van Holt 2012). As the management system restricts artisanal fishers to harvesting in only their own local area, they have little ability to adopt to changes in shellfish quality (by fishing elsewhere). Limited responses include harvesting in unassigned areas (where *loco* populations are overfished) outside their management area, going further offshore to fish for other species, or leave the fishery altogether (Van Holt 2012). In the Mehuin region of Chile, an area with poor quality *locos*, fishers are leaving the fishery due to the low-quality resource, and conflict with the Celco Arauco pulp mill. Unfortunately, tree plantations are likely to increase in southern Chile as many areas near current plantations are already cleared or in the early stages of forest establishment (Van Holt 2009). Van Holt et al. (2012) concluded that "*the development of tree plantations within watersheds at or near the land-sea interface results in increased chlorophyll-a concentrations in adjacent nearshore waters (eutrophication) and promotes the growth of fouling organisms (suspension feeding endobionts and epibionts) that, in turn, affect the biological condition of locos*", and that "*that more integrated approaches to natural resource management are needed, with socio-economic trade-offs associated with specific land uses explicitly identified and incorporated into decision making processes".*

Saunders et al. (2012) used modelling to quantify the link between terrestrial sediment runoff and a downstream coastal marine ecosystem and contrasted the cost-effectiveness of marine and land-based conservation actions. They assessed the coral reefs of Fiji, by estimating their relative condition as influenced by watershed-based pollution (using a proxy for sediment, nutrient, and pollutant delivery) and fishing. The cost-effectiveness of protecting forests was evaluated, and it was found that investments to protect forests had rapidly declining returns (as measured by relative reef condition). For example, the protection of 2% of forest in one area, was almost 500 times more beneficial than protecting 2% in a different area. Spatial prioritisation of spending was considered essential.

For the scenarios evaluated, relative coral reef condition could be improved by 8–58% if all remnant forest in Fiji was protected rather than deforested. A determination was made of the priority of each coral reef for implementing a marine protected area, if all remnant forest was protected for conservation (Figure 26). These results were to be used by the Fiji Protected Area Committee as they established a national protected area network aiming to protect 20% of the land and 30% of the inshore waters.

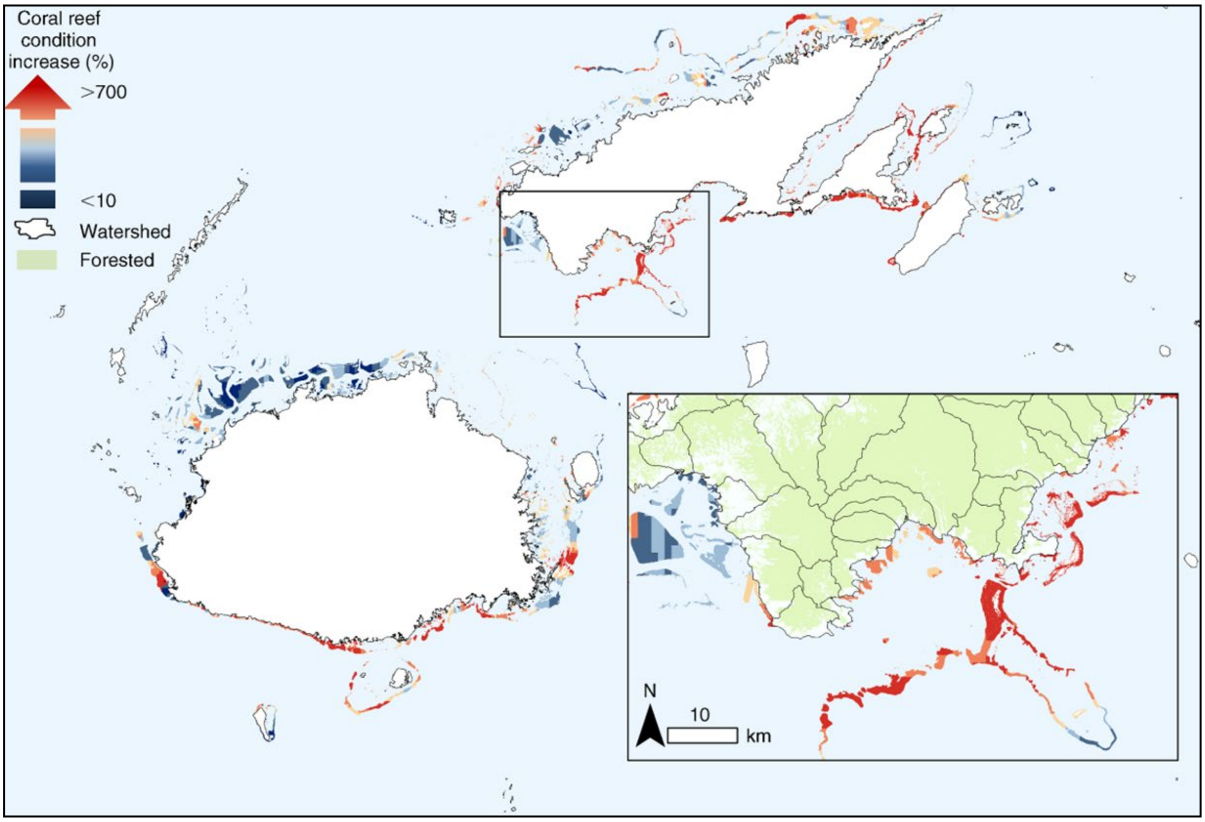


Figure 26: Percentage increase in coral reef condition when all forests are protected rather than cleared. Results are reported for each 1-km^2 coral reef pixel. The inset shows results from southern Bua Province on the island of Vanua Levu in greater detail. (Source: figure 4 of Saunders et al. 2012.)

Brown et al. (2019) reviewed different modelling studies to generate a guide to modelling priorities for managing land-based impacts on coastal ecosystems. The demand for quantitative models to support evidence-based planning is increasing (Carroll et al. 2012), with quantitative models aiding planning, as they allow predicted outcomes for different ecological or economic responses to land-use change. Linking land uses to fisheries requires an interdisciplinary approach, as models must cover a wide range of themes, including land-use change, pollutant paths, physical and chemical oceanography, fisheries ecology, economics, and social science. This is a significant challenge for the typically small teams running on tight time schedules that develop land-sea plans (Brown et al. 2017a). Development of decision support tools for these small teams must balance the improved precision of complex process-orientated models against the additional development time need to build such models. To date, the integrated coastal zone management literature has provided mostly conceptual models (e.g., Stoms et al. 2005). Conceptual models can be used to suggest precautionary management guidelines or rank environmental state on ordinal scales, but they are weak in their arbitrary data analysis and may misrepresent system dynamic uncertainty (Game et al. 2013). In contrast, incorporating models on land-use impacts to fisheries into quantitative planning frameworks provides a transparent and repeatable approach (Game et al. 2013).

Brown et al. (2019) searched the literature for peer-reviewed examples of spatial land-sea plans and found 14 examples (Table 6). In general, existing planning approaches could be divided into either threat-based or outcome-based categories (Giakoumi et al. 2015). Threat-based approaches aimed to reduce the amount of threat to marine ecosystem(s) and species (e.g., Klein et al. 2010); while outcome-based approaches aimed to maintain or improve the state (e.g., health) of marine ecosystem(s) or species through a reduction in threats (e.g., Klein et al. 2012). Brown et al. (2019) advocated moving to outcome-based approaches, arguing that they are directly connected to the ultimate objectives of planning and avoid nominal variables that may have unclear and unquantified relationships with a planner's objectives. Most effort has focused on threat-based approaches, probably due to a lack of data and/or the greater level of modelling needed for outcome-based approaches.

Implicit in many threat-based approaches was the assumption that threats related linearly to the desired outcomes (i.e., the more threat was reduced, the greater the likelihood of marine ecosystem transition to the desired state). The linear relationship assumption could be wrong in many circumstances; for instance, where ecological or economic tipping points drove nonlinear change in fisheries (Selkoe et al. 2015). Outcome-based approaches avoid such assumptions. The main barriers for the use of outcome-based approaches in land-sea planning are reliable models and data that can predict how the type and quantity of threats impact a marine ecosystem. Brown et al. (2019) provided detailed discussion on a range of modelling areas: including drivers of land-use change; human activities that cause land-use change; sediment and nutrient run-off from land; the dispersion and transformation of pollutants in the ocean; the response of ecosystems and fish populations to pollution; and the economic, social, and human responses to ecological change. Those discussions are too detailed and wide-ranging to summarise here, and the reader is encouraged to read Brown et al. (2019) paper fully.

Table 6: The 14 quantitative land-use studies that linked land-use change to coastal ecosystems and fisheries, as reviewed by Brown et al. (2019). Shaded boxes indicate steps where a specific quantitative model was used, empty boxes indicate that no quantitative model was used, though that step may have been considered conceptually. (Source: table 1, Brown et al. 2019.)

Example Coastal Ecosystems	Drivers of land use change	Human activities that cause land-use change	Run-off of pollutants	Disperson of pollutants in the ocean	Ecosystem response	Socio-economic response	Planning	References
New Ireland Province, Papua New Guinea		■	■	■	■		■	Tulloch et al. 2016
Hawaii groundwater run-off to reefs		■	■	■	■			Delevaux et al. 2018
Gulf of California marine ecosystems		■	■	■			■	Alvarez-Romero et al. 2015
Hawaii run-off to coral reefs		■	■				■	Oleson et al 2017
Guam coral reefs			■	■	■	■	■	Weijernan et al 2016
Moreton Bay seagrass			■	■	■		■	Saunders et al. 2017
Fiji coral reefs			■	■	■		■	Klein et al. 2012
Fiji coral reefs			■	■	■			Brown et al. 2017a,b
Great Barrier Reef and its catchments				■				Wolff et al. 2018
Southern Chile, Loco fishery					■	■		van Hott et al. 2012
Gulf of Mexico shrimp fishery					■	■	■	Huang & Smith 2011
Western Phillipines coral reefs					■		■	Melbourne-Thomas et al. 2011

The authors made several future research suggestions. These were:

1. Where possible, researchers should attempt to model the outcomes of land-use change for coastal systems instead of using threat indices, even if the modelling was static (e.g., Delevaux et al. 2018).
2. A challenge for planning with static or dynamic models is the consideration of uncertainty at any stage of the linked land-sea process. Methods are needed that can propagate uncertainty, so that key uncertainties can be quantified.
3. The effects of extreme weather events on run-off and coastal fisheries are poorly understood. More work is needed to understand how extreme events affect coastal fisheries indirectly by temporary changes in water quality, and how the timing and severity of events may change under different climate change scenarios.
4. Models that can consider dynamic feedbacks in socio-economic systems like fisher behavioural responses to changes in water quality are needed. Dynamic feedbacks may render plans ineffective or may be supported by planners where they enhance the capability of people to adapt to changes in fisheries (e.g., Van Holt 2012). These dynamic feed-backs could also

consider how large-scale drivers, like climate change and globalisation of economies, impact on the effectiveness of land-sea plans.

5. Management plans are often developed on relatively short time scales with limited funds for future research, so the development of precise models that link land-sea processes may not be an effective investment of time and funds. Modellers can help inform by reporting on development time required to achieve models of differing complexity and precision. Models can also be used to inform on rules of thumb that can be used to aid planning in other data-poor situations. For instance, geographical context can be used to decide whether land or sea actions are more cost effective for achieving marine habitats conservation (Saunders et al. 2017). Similar rules of thumb are needed for the socio-economic impacts of run-off.
6. Actively involving a wide range of stakeholders (such as industry, local fishers, NGOs, government departments) in planning processes is a fundamental step in integrated land-sea management. Engaging stakeholders by asking them to contribute to model development may help fill in data-gaps and increase by-in to model results. For instance, participatory mapping exercises have proven to be highly effective at extracting fine scale spatial information on current and future land based threats to marine systems (e.g., Game et al. 2011).
7. While environmental NGOs may have the skills and relationships needed to facilitate participatory planning processes in data-poor countries, they rarely have the expertise needed to develop dynamical models that link land to sea processes. One way to address this gap is to have external ecological modellers become stakeholders in planning processes.

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

6.1 Coastal fisheries

New Zealand’s extensive coastline, encompassing more than 1600 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 has been at the Whangārei Harbour entrance, Northland (Mair Bank, closed since 2014), while commercial cockle fisheries existed inside Whangārei Harbour (Snake Bay, closed since 2012), further south in Tasman/Golden bays (Pakawau Beach, Tapu Bay-Riwaka, Ferry Point), and in Otago (Papanui, Waitati, Purakanui inlets and Otago Harbour). Total 2018–19 annual cockle and pipi catches were 1095 and 135 t respectively (all catches given are from Fisheries New Zealand 2020). On exposed beaches, tuatua (*Paphies subtriangulata*, *Paphies donacina*) replace cockles and pipi, with a small commercial fishery (currently not operating) in 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 focused around east Northland (242 t) (fishery closed in 2022), Coromandel (includes greater Hauraki Gulf) (471 t) (fishery closed in 2022), and the Nelson/Marlborough areas (closed since 2018) (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), pāua (*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 – 1939 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 (*Chrysophrys auratus*) dominate commercial catches (6328 t), along with a related fishery for trevally (*Pseudocaranx dentex*) (2987 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*) (3817 t), John dory (*Zeus faber*) (815 t), kahawai (*Arripis trutta*) (2500 t), and tarakihi (*Nemadactylus macropterus*) (5729 t). Further south, on shallow rocky reefs and on biogenic (living) reefs, blue cod (*Parapercis colias*) become important (1844 t), along with blue moki (*Latridopsis ciliaris*) (543 t), while soft sediment associated species include red cod (*Pseudophycis bachus*) (5551 t) and elephantfish (*Callorhinchus milii*) (1148 t). School shark (*Galeorhinus galeus*) (3719 t) and rig (*Mustelus lenticulatus*) (1362 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*) (10 300 t), and trevally. In the high energy environment of Foveaux Strait, the important Bluff oyster (*Ostrea chilensis*) fishery still operates (7.5 million oysters quota 2022), albeit at a level much reduced from catches in past decades, along with a smaller fishery in Nelson/Marlborough (132 t).

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.

6.2 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 various 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 (closed since 2018).

6.2.1 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 Whangārei Harbour, while cockles are collected from Whangārei Harbour (Snake Bank), Golden/Tasman bays, and in Papanui and Waitahi estuaries, Otago (Fisheries New Zealand 2020). Despite the present-day widespread abundance of these species, 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, with some gaps (e.g., between 2006 and 2009). A series of Fisheries New Zealand 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, Pawley 2011, Berkenbusch et al. 2015, Berkenbusch & Neubauer 2015, 2016, 2017, 2018, 2019, 2020), 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 focused on the estimation of numbers and biomass of the populations over time and human harvest levels. Little associated temporal environmental information has been recorded, although sediment scrapes have been collected and archived for more recent surveys. 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*".

Neubauer et al. (2015) analysed part of this series using a Bayesian state-space model that simultaneously analysed monitoring data from 21 incomplete survey time series (i.e., 21 different beaches), between 1999–2000 and 2014–15. Several Coromandel sites (Whangapoua, Tairua, and Whangamatā harbours) showed strong but smooth declines in large-size cockle densities, whereas several Hauraki Gulf sites (Eastern Beach, Cockle Bay, and Kawakawa Bay West) showed distinct increases in cockle densities. Beaches closed to harvesting were associated with a clear positive change in their trend estimates. This included posterior mean estimates of overall population growth rates of 17% at Eastern Beach and 24% at Whangateau Harbour, in contrast to negative estimated growth rates of 6% to 9% declines per year, respectively, prior to being closed to harvesting. Neubauer et al. (2015) concluded that beach closures are very likely to enable population recovery.

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 3000 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 27 (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 adverse 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, and the reproductive status of adult cockles and pipi was also negatively affected (Gibbs & Hewitt 2004).

Anderson et al. (2019) looked at the ability of New Zealand cockles to migrate up out of marine sediment deposition, using tank experiments to mimic real-world sedimentation events. Cockles were found to be highly mobile (upward burrowers, resurfacing within days from burial under 2 to 25 cm of sediment), if buried as undisturbed animals (in their normal orientation with siphons facing up). They were also robust to daily burials under 2 cm of sediment. In contrast, when they were disturbed in their normal orientation before burial, 5 to 10 cm of sediment was sufficient to significantly reduce rates of reappearance at the sediment surface, with adults having higher failure/loss rates than sub-adults.

Many relic beds of dead cockles are found buried under 50 cm or more of soft muddy sediments in New Zealand estuaries. Their ability to return to the surface when buried under sediment in tank-based settings suggests that other processes were also likely to be in play during these historical burial events. It is likely that during large floods, very fine muds and silts carried in freshwater down to estuaries may form a fine slurry, whose semi-liquid form prevents animals from being able to burrow through upwards to the sediment/water interface. Other experimental work has shown that thin layers of clay (3, 6, 9 cm deep) deposited in a freshwater slurry (simulating terrestrial runoff) prevented individual cockles from resurfacing (Norkko et al. 2002, Lohrer et al. 2004). Oxygen depletion might also be an issue with fine terrestrial sediments likely to be carrying associated loads of organic debris and detritus.

Comparison of present day and historical cockle populations in the Bay of Islands has further documented the extensive spatial loss of cockle beds (Booth 2020). Human settlement in this area started around 1300 AD (Robinson et al. 2019), with early European accounts indicating a large Māori population (> 10 000) in the Bay of Islands at the time of the first European arrivals (1769, then 1772) (Booth 2017). As with other regions, Māori used fire to clear much of the forest cover, to promote the growth of starch-rich bracken (*Pteridium esculentum*). Root networks remained and slowed sediment erosion from the area catchments, with relatively low increases in SAR (Swales et al. 2012). Following European settlement and widespread land clearance for farming and later pine plantations and citrus orchards, soil erosion rates increased significantly. This was especially so for steeper soft-rock hill-country soils, which are highly vulnerable to erosion and landslides (Wilmshurst 1997). Bay of Islands SARs are now 10–20 times higher than before European settlement (Swales et al. 2012).

Booth (2020) used the cockle population of mid Kerikeri Inlet as an example of loss over time, attributed to sedimentation. The inlet's shoreline has extensive 'pre-contact' cockle shell middens, dominated by shells of 30–55 mm width, which were historically mined to produce burnt lime, as well as being trucked both whole and machine-crushed to nearby farm properties. By the 1940–60s local families were still reliant on and harvesting large cockles here and elsewhere in the Bay of Islands (oral accounts given by Booth 2020), but now in the present day there is little cockle gathering in Kerikeri Inlet, extensive areas of deep mud are present with no cockles, and while cockles are still abundant in some places, they are much smaller in size and do not grow to sizes that are favoured by recreational gatherers (Hartill et al. 2005). Even the largest cockles present do not match the shell sizes seen in the adjacent middens. This suggests that while ongoing 'good' cockle recruitment continues, something has changed so that cockles no longer grow to the large sizes seen historically. This pattern is seen in many other cockle populations around New Zealand where increased land-based sedimentation has occurred, suggesting reduced cockle growth rates and/or increased mortality rates at larger adult sizes. For example, further south at Snake Bank in the lower Whangārei Harbour, the once abundant cockle population and associated fishery (which was closed in 2015) had an asymptotic upper cockle size of 35 mm; this contrasts with many cockles in nearby middens that are 50 mm or larger.

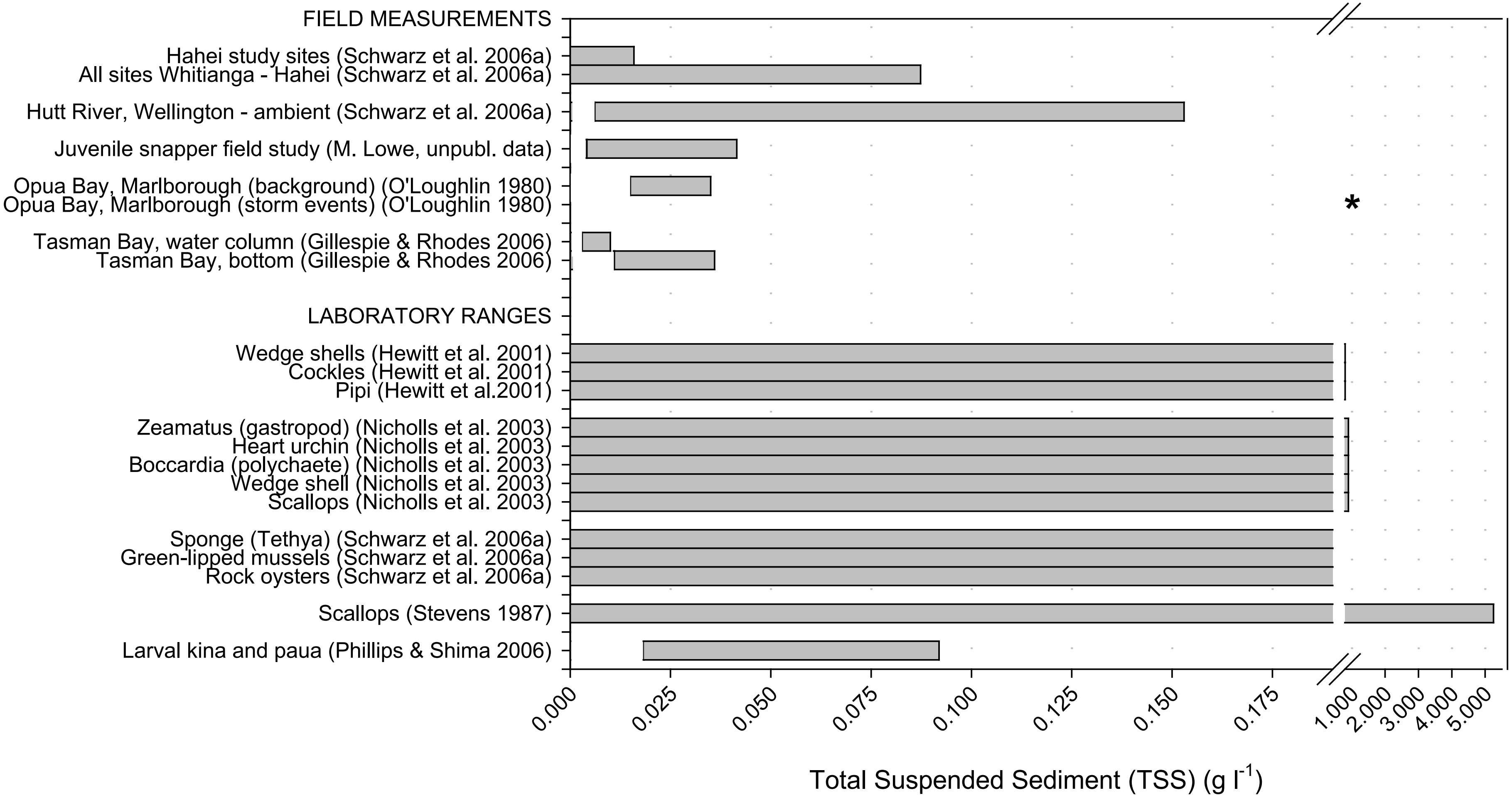


Figure 27: Summary of total suspended sediment concentrations from a range of New Zealand studies discussed in this report (adapted from Schwarz et al. 2006a). *, storm event.

6.2.2 Tuatua (*Paphies subtriangulata, P. donacina*), toheroa (*P. ventricosa*), and other 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 review of possible drivers of toheroa population declines was completed in 2013, which identified potential contributing factors to include food availability, climate and weather, sand smothering/sediment instability, toxic algal blooms, predation, harvesting, vehicle impacts, and land-use change (Williams et al. 2013). Available data allowed for a desk-top exploration of climate and weather, toxic algal blooms, vehicle impacts, and land-use change. There was an association between toheroa mass mortality events and negative values of the Trenberth climate index Z at the time of the event, indicative of easterly zonal flow. This corroborated anecdotal observations that mass mortality events often coincided with easterly winds.

There was strong evidence that the use of vehicles on beaches damaged toheroa. Most vehicles drove on the mid to high tide area of the beach, where the densest beds of adult and juvenile toheroa, respectively, occurred, with both size groups appearing vulnerable to vehicle traffic. Toheroa beds were associated with freshwater seepage areas (SGD) on the beach, and changes in land use adjacent to the beach was suggested to affect the presence of these seeps. An examination of land-use changes for Ninety Mile and Dargaville beaches found that forests (pine) have replaced ‘open’ dune systems along Ninety Mile Beach, which may have reduced freshwater SGD rates along the beach [Note: there is an unverified historical account that Ninety Mile Beach may have once been fringed by pōhutukawa trees, many of which were cut down to fuel the short-lived toheroa canneries that once operated there]. The lands behind Dargaville Beach had a lesser change in land use; toheroa populations are in the present day more abundant on Dargaville Beach than on Ninety Mile Beach.

Investigations of the Aupōuri aquifer have suggested that the large-scale planting of pine forests since the 1970s has reduced recharge of the aquifer (possibly by more than 50%) (HydroGeo Solutions 2000, Cameron et al. 2001) and, by association, surface and groundwater freshwater delivery to Ninety Mile Beach. Water yield tends to decrease as pine forest cover in a catchment increases, with small to medium size catchments largely converted to forest from previous pasture or tussock grassland having reduced water yields of up to 55% (Fahey & Rowe 1992). With New Zealand currently experiencing a surge in the conversion of pastures to extensive pine forests, including for carbon credits, increasing pressures on aquifer recharge and associated SGD are highly likely.

For toheroa populations, this may lead to reduced toheroa populations access to beach areas with freshwater seepage. The loss of these wetter (and possibly cooler and elevated nutrient concentration) beach areas has possibly cascaded through into negative effects on shellfish populations.

Surf clam assemblages (multiple 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. However, recent mortality events have been observed, although their extent and cause are unknown.

6.2.3 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, and as foraging areas for adult fish.

Mussel fishing as an industry was first noted in the New Zealand Marine Department Annual Report for 1913–14, where mussel curing and canning factories were noted as present at Tapu and Thames. By the 1920s a larger fishery started to develop, with the earliest beds worked inside Coromandel Harbour and in the island channels adjacent to the entrance (Paul 2012). Paul (2012) provides a comprehensive review of the life span of the fishery, building on earlier works including Reid (1968) and Greenway (1969). The fishery was dominated by the Strongman and Gundlock families. Throughout the 40 years of the fishery, 2–4 vessels operated full-time, with some additional casual boats, landing a high of 40–900 sacks of mussels in 1961 (estimated at 15 million mussels). Despite warning signs of over-exploitation and other stock issues over several decades, 2600 t were landed (green weight) in 1960 and almost 2800 t in 1961 (the highest annual landing recorded). After this, annual catches started to drop away rapidly, even as searching time increased; with 1650 t in 1963, 850 t in 1964, and 100 t in 1966 (Paul 2012). By 1966 the fishery had collapsed (Greenway 1969). Full-time dredging ended in 1967, and landings in 1969. Beds were serially depleted; as the populations around Coromandel and the eastern firth shoreline declined, effort turned to the southern and western areas, then to Ponui Island, and finally to random searching over the whole area (Greenway 1969) (Figure 28). The prevailing theory at the time seems to have been that the populations were unsustainably fished down, but at the same time dredging had removed settlement surfaces for newly recruiting mussels, which were extracted as part of the mussel/byssus matrix. It was thought that a spatial refuge remained for populations on rocky reef areas, but that this was under threat from increasing recreational harvesting (Greenway 1969).

Acoustic and towed camera surveys in 2001 and 2002 of the Firth of Thames and the western inner gulf, respectively, using single beam acoustic (both surveys) and sidescan sonar (2001 survey only) methods, found no sign of bed recovery (only a very few mussel clumps were seen in the Firth of Thames, largest about 0.5 m^2) (Morrison et al. 2002, 2003). Investigation of an inner-mid firth location based on local knowledge, targeted with multibeam sonar, reported a larger bed (D. Immenga, University of Waikato, unpublished data, not sighted). Further direct searching by McLeod et al. (2012) found only a few scattered beds/remnants totalling about 0.64 km^2. Two of these were adjacent to mussel farms and were likely formed by live mussel drops: one in the outer west Firth of Thames off Waimango Point in 5 m water depth, the other in the Man Of War/ Ponui Passage off east Waiheke Island, in 20 m water depth. One other was in 0.5 m water depth in one of the Weiti River Estuary channels (an estuary adjacent to the Whangaparāoa Peninsula), and the final one in the Okiwa Estuary in 0.5 m water depth, on the northeastern coast of Great Barrier/Aotea Island.

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 present-day 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, unpublished 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 surfaces 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, 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.

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, to the point that they are now rare in the Wadden Sea (Wolff 2000) and considered to be threatened in the United Kingdom (Hiscock et al. 2005).

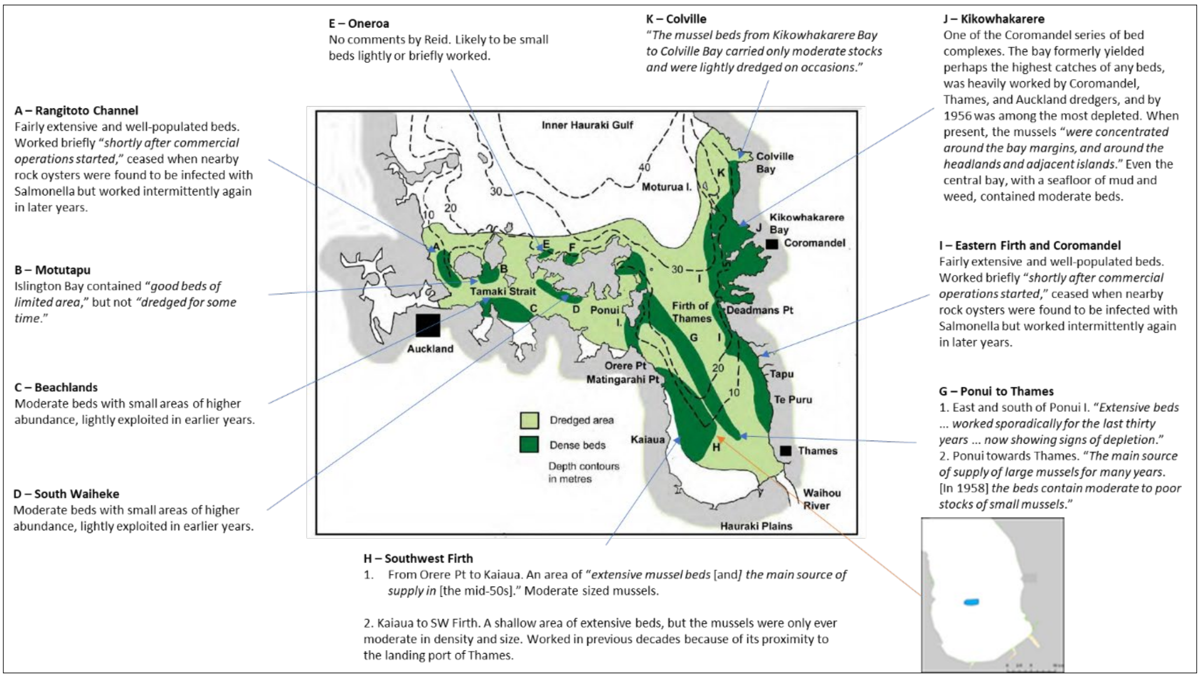
McLeod et al. (2014) looked at the role of subtidal green-lipped mussel beds on soft sediments as a biogenic habitat for other species in the Hauraki Gulf. Using cores, the invertebrate fauna on and under mussel patches, and for adjacent bare sediment controls, was quantified. Invertebrate densities of the mussel patches (epifaunal and infaunal) were 2 to 8 times greater than the adjacent bare sediments. Associated biomass values were 7 times higher. Species richness was also higher, especially of small crustaceans (0.5–5.6 mm) including calanoid copepods, caridean, cumacean and mysid shrimps, paguroidea (hermit crabs), porcellanidae (half crabs), pycnogonidia (sea spiders), and tanaidecea; all of which were absent in adjacent bare areas. Larger crustaceans (8.0–22.4 mm) were also much more abundant in mussel habitats, with high densities of *Petrolishes elongates* (blue half-crab) and *Halicarcinus innominatus* (pill box crab) at some sites. Such strong associations between mussels and high densities of crustaceans have also been documented in a number of international studies (e.g., Dittman 1990, Moksnes et al. 1998, Ragnarsson & Raffaelli 1999, Moksnes 2002, Beadman et al. 2004, Lindsey et al. 2006), as well as the broader assemblage contrasts between mussel and adjacent non-mussel habitats (Ragnarsson & Raffaelli 1999, Duarte et al. 2006, Commito et al. 2008).

McLeod et al. (2014) calculated the potential loss of macrofaunal invertebrate productivity associated with the historical loss of 500 km^2 of Hauraki Gulf soft sediment mussel beds (Figure 28, pre-1958 estimate, Reid 1968. Using two estimates of historical mussel densities of 1.5 per m^2 and 120 per m^2 (see above), the total small mobile invertebrate loss estimates were 370 and 33 000 tons Ash-Free-Dry-

Weight y^{-1} (note: this does not include larger mobile invertebrates such as predatory starfish and wandering anemones).

MacLeod et al. (2014) also sampled the small fish above mussel patches and adjacent bare sediment using small 1-m^2 drop nets operated by divers. The fish catch was dominated by mottled triplefins (*Grahamina capito*), clingfish (*Trachelochismus melobesia*), and small spotties (*N. celidotus*). Overall small fish densities ranged from 2 to 10 individuals per square metre across the three locations sampled and were around ten times greater than densities on adjacent bare sediments. It is important to note the limitations of such diver-operated small drop nets in sampling juvenile/small fishes; more mobile species such as juvenile snapper and trevally will flee with the approach of divers, and, in this instance, the timing of sampling (October, May) did not coincide with the presence of very small juveniles of these species in northeastern New Zealand. Other small relatively abundant fish species associated with epifaunal habitat-forming species that may grow on established mussel beds (e.g., bastard red cod and *Callyspongia* sponge spp.; juvenile leatherjackets with sponges and macroalgae) were also not sampled.

Given the large secondary productivity declines, changes in infaunal size distributions, and the loss of crustacean components going from mussel beds to bare sediments, a strong cascading effect into epibenthic carnivores including fish (e.g., snapper) was highly likely. Using the two lost productivity estimates given above (370 and 33 000 tons Ash-Free-Dry-Weight y^{-1}), McLeod (2009) estimated that the small mobile invertebrate productivity associated with the pre-1958 mussel reefs could have supported an additional biomass of between 200 and 16 000 tons y^{-1} of predatory fish above those able to be supported by 'bare' sediment areas, which replaced the reefs from the late 1960s onwards. The extra production supported by the current day Waimangu Point mussel bed (640 000 m^2) was estimated to be 20 tons y^{-1}.



E – Oneroa
No comments by Reid. Likely to be small beds lightly or briefly worked.

K – Colville
"The mussel beds from Kikowhakarere Bay to Colville Bay carried only moderate stocks and were lightly dredged on occasions."

J – Kikowhakarere
One of the Coromandel series of bed complexes. The bay formerly yielded perhaps the highest catches of any beds, was heavily worked by Coromandel, Thames, and Auckland dredgers, and by 1956 was among the most depleted. When present, the mussels *"were concentrated around the bay margins, and around the headlands and adjacent islands."* Even the central bay, with a seafloor of mud and weed, contained moderate beds.

A – Rangitoto Channel
Fairly extensive and well-populated beds. Worked briefly *"shortly after commercial operations started,"* ceased when nearby rock oysters were found to be infected with Salmonella but worked intermittently again in later years.

B – Motutapu
Islington Bay contained *"good beds of limited area,"* but not *"dredged for some time."*

C – Beachlands
Moderate beds with small areas of higher abundance, lightly exploited in earlier years.

D – South Waiheke
Moderate beds with small areas of higher abundance, lightly exploited in earlier years.

I – Eastern Firth and Coromandel
Fairly extensive and well-populated beds. Worked briefly *"shortly after commercial operations started,"* ceased when nearby rock oysters were found to be infected with Salmonella but worked intermittently again in later years.

G – Ponui to Thames
1. East and south of Ponui I. *"Extensive beds … worked sporadically for the last thirty years … now showing signs of depletion."*
2. Ponui towards Thames. *"The main source of supply of large mussels for many years.* [In 1958] *the beds contain moderate to poor stocks of small mussels."*

H – Southwest Firth
1. From Orere Pt to Kaiaua. An area of *"extensive mussel beds* [and] *the main source of supply in* [the mid-50s]." Moderate sized mussels.

2. Kaiaua to SW Firth. A shallow area of extensive beds, but the mussels were only ever moderate in density and size. Worked in previous decades because of its proximity to the landing port of Thames.

Figure 28: Map of historical green-lipped mussel distribution in the Hauraki Gulf, as created by Reid (1968), with associated notes on the fishery history of the (ten) individual main mussel beds. Also shown in the lower right smaller figure is the location of the mussel patch mapped by D. Immenga in about 2003. Source: adapted from figure 5 of Paul (2012), who redrew the map from Reid (1968), and added depth contours; the text is also from Reid (1968), as paraphrased in table 1 of Paul (2012); adapted from figure 2 2 of Thompson et al. (2020).

Considerable efforts are now being made to restore green-lipped mussel beds both in the Hauraki Gulf and in Marlborough Sounds, where large wild beds were also lost. Initial pioneering MSc thesis work was done by McLeod (2009) and subsequently published by McLeod et al. (2012). Adult mussels were transplanted into small cages on the seafloor at three locations along the western side of the Firth of Thames that fell along a turbidity gradient (average visibility 0.8–4.7 m). After 500 days, 68% of the mussels were still alive and had grown an average of 19 mm in length. Mussels at the least turbid site were in better condition than those from the most turbid site. Deployment of settlement surfaces (the live mussels, adjacent dead mussel shell, and spat collecting materials) did not return any new mussel recruits. They concluded that the lack of recovery of mussel reefs in the firth was due to low recruitment and survivorship of mussels, and that the "*restoration of mussel reefs and the ecosystem services that they provide may therefore be possible*" (McLeod et al. 2012).

Subsequent to McLeod's work, a group known as Revive Our Gulf/The Mussel Restoration Trust was set up to work to restore the green-lipped mussel beds of the Hauraki Gulf (http://www.reviveourgulf.org.nz/), and a number of reports and science papers have since been published looking at how mussel beds might be restored (e.g., Sim-Smith 2015, Sim-Smith & Kelly 2015, 2016, Wilcox & Jeffs 2017, 2019, Wilcox et al. 2018, 2020). Restoration sits outside of the scope of this present review, but a Fisheries New Zealand review looking at the potential for habitat restoration in the Hauraki Gulf (Morrison 2021) provides a summary of those later mussel restoration papers.

6.2.4 Scallops (*Pecten noveazelandiae*)

Scallops are an important and highly valued coastal shellfish species. They tend to occur in aggregations (beds) often associated with coarser seafloor 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; although current Fisheries New Zealand projects are looking at these relationships (no results available at the time of report writing). 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 in that harbour are currently under a rāhui due to perceived low numbers). Historically, abundant scallop beds in Whangārei Harbour were associated with subtidal seagrass meadows, which disappeared in the 1960s along with the seagrass (see Section 6.5.1, Morrison 2003). Casual observations of the inner Kawau Bay area, in particular around Rabbit and Tarahiki 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 in 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 the scallop gills being blocked up with fine-grained particles (*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 their 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 m) 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, whereas 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 phenomenon 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 were halted in 2014, with the suggestion that the quantity and quality of SPM available for benthic suspension feeders may be a major contributing factor (Gillespie & Rhodes 2006).

6.2.5 Pāua (*Haliotis iris*) and kina (*Evechinus chloroticus*)

Pāua and kina are highly valued, with the first supporting very important commercial fisheries, as well as customary and recreational harvest. Little quantitative information appears to exist on how land-based effects directly impact on adult populations, although gross alteration of some inshore reef habitats by sedimentation seems to be having strong negative effects on local fisheries (e.g., Māhia Peninsula, Miller et al. 2007). Some work has been undertaken on the spawning, larval, and settlement phases of these shellfish.

Miller et al. (2014) used tank experiments to assess the effects of different suspended sediment levels on the fertilisation success of kina, using levels like those encountered in the field. They used both generalised linear mixed models (GLMMs) and hierarchical Bayesian (HB) regression to model the experimental data. Both approaches showed a significant decrease in fertilisation success with increasing suspended sediment levels. They noted that their model estimate uncertainties were less for the Bayesian model and suggested thus approach had advantages where only sparse datasets were available.

Phillips & Shima (2006) looked at the effects of suspended sediments on kina and pāua 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. Pāua 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. Pāua 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 & Shima (2006) noted that natural cohorts of pāua 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.

Glockner-Fagetti & Phillips (2019) experimentally assessed the combined stress of low salinity and suspended or deposited sediments on larval and post-larval development of kina. Larvae were exposed to low salinity (28, 32 ppt) and suspended sediments (40, 80 mg l^{-1}), in a fully factorial design, for four days, then returned to control conditions (36 ppt, 0 mg l^{-1}). Four weeks later, competent larvae from the different treatments were placed in bowls and allowed to settle. Half of the bowls were allocated a treatment of covering the bowl bottoms with fine sediments, to assess deposited sediment effects. Seven days after fertilisation, larval development was reduced in the lowest salinity and suspended sediment treatments, but these differences disappeared by Day 21. Larval survival was unaffected. However, settlement success and juvenile development were reduced both by larval exposure to the lowest salinity and the presence of deposited sediments. At 28 days post-settlement, there were no surviving juveniles from the lowest salinity and highest suspended sediment treatments. It was concluded that even short exposure to runoff during early development could have strong consequences for later larval settlement and juvenile performance, with flow-on effects to subsequent recruitment to adult populations.

Rouchon & Phillips (2017) investigated the toxicity of three major metal pollutants (copper, lead, and zinc) often found in coastal waters near urban and industrial areas on kina larvae. Larval development assays were conducted for 72 h. Median effective concentrations (EC50) for normal larval development for individual metals were 5.4 µg/L Cu, 52.2 µg/L Pb, and 27.7 µg/L Zn (EC50 is the level at which 50% of tested individuals show adverse effects). Kina were more sensitive to copper and zinc than most other echinoid species that have been tested. The effects of metal mixtures were analysed using a toxic unit (TU) approach. Cu + Zn and Cu + Pb had a less-than-additive effect on kina larval development. By contrast, Zn + Pb was strictly additive. None of the interactions were strong, with sums of TU ranging from 1.00 to 1.85. These results were concluded to support water quality thresholds that use strictadditivity approaches to set water quality criteria (as recommended in New Zealand and Australia, ANZECC 2000).

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).

6.2.6 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 the past by Fisheries New Zealand working groups (K. Sullivan, previously Ministry of Fisheries, pers. comm. 2009), 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 fell from a peak of 16 572 tons in 1951, to about 2300 tons in the early 2000s (an 86% decline), while Namibian catches fell from a peak of 9 189 tons in 1955 to about 290 tons (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 eliminating benthic organisms between 5 m depth and the mid-littoral zone (Branch et al. 1990 from 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 Hawke's Bay.

6.2.7 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.

6.3 Freshwater finfish

In New Zealand, evidence is emerging of sub-lethal concentrations of freshwater suspended sediments affecting the upstream migrations of native freshwater fish species, of which 70% are diadromous (McDowall 1990). Feeding experiments on migrant banded kōkopu (*Galaxias 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 kōkopu, ultimately resulting in reduced recruitment of juveniles in turbid rivers (Richardson et al. 2001). Other field studies have noted reduced densities of juvenile kōaro (*G. brevipinnis*), īnanga (*G. maculatus*), and banded kōkopu in highly turbid waters following floods (McDowall & Eldon 1980) and within catchments containing high production pasture compared with forested streams (Hanchet 1990, Minns 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 1990). Reports of prodigious whitebait catches declining drastically over the past 100 years have been documented by Phillips (1924a cited by McDowall 1990) with reported "cartloads" of whitebait coming from the Hutt River in the 1880s. Catches such as this were common from all over New Zealand.

6.4 Snapper (*Chrysophrys auratus*) and grey mullet (*Mugil cephalus*) juveniles

We have limited information on the direct effects of stressors such as sedimentation on fish in the New Zealand marine context. Lowe et al. (2015) looked at the impacts of suspended sediment loads on juvenile snapper (*C. auratus*). Juvenile snapper less than one year of age (50–100 mm fork length) were collected 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, 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 to the Waitematā Harbour as the least 'pristine' having the lowest (Figure 29). In addition, much 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*, whereas in the more turbid estuaries (Manukau, Kaipara, Waitematā) diet was dominated by benthic prey. These findings suggested 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 cause a reduction in the actual zooplankton assemblages available due to changing environmental conditions.

A subsequent study on juvenile snapper growth rates (using daily growth rings) by Stewart (2018) compared growth rates from subtidal seagrass meadows in East Northland (Pārengarenga, Rangaunu, and Whangārei harbours) with nearby non-vegetated juvenile snapper nursery areas. Juvenile snapper from seagrass habitats grew significantly faster and had higher condition indices (weight-at-age). At 40 days of age post-settlement, juvenile snapper from subtidal seagrass areas weighed 1.45 times more than non-seagrass fish; this increased to 1.87 times by 70 days age (Stewart 2018). Land-based sedimentation is considered to a major driver in the loss of historically extensive subtidal seagrass meadows in northern New Zealand and, by extension, substantial loss of fish productivity.

Juvenile grey mullet (*Mugil cephalus*) appear to be obligate on estuaries as nurseries in New Zealand (Morrison et al. 2016). Mohd-Zain (2013) looked at the relationship between juvenile grey mullet growth rates (using daily rings) and the environmental conditions (derived from an estuary database and models) in 14 northern New Zealand estuaries (selected to encompass environmental gradients). Ageing using daily rings proved to be difficult for this species, and fish ages were weakly correlated with lengths for many of the estuaries. The average growth rate of juveniles from each estuary was obtained from the slope of a regression of length on age (with the line constrained to pass through the origin) and ranged from 0.18 to 0.36 mm d^{-1}. Despite the uncertainties associated with ageing fish, average growth rate was moderately correlated with a number of environmental variables at the level of estuary (Figure 30). The best explanatory variable was the percentage of intertidal area occupied by mangroves, which accounted for 50% of variation in average growth rate, with fish growing slower as mangrove coverage increased. Rates of sediment and nutrient input were also negatively correlated with fish growth ($R^2 = c$ 0.4). Lowe (2013) found low density and productivity values for invertebrate prey utilised by estuarine fish assemblages (including juvenile grey mullet) in mangrove forests, compared with seagrass meadows and bare intertidal flats (all in northern New Zealand).

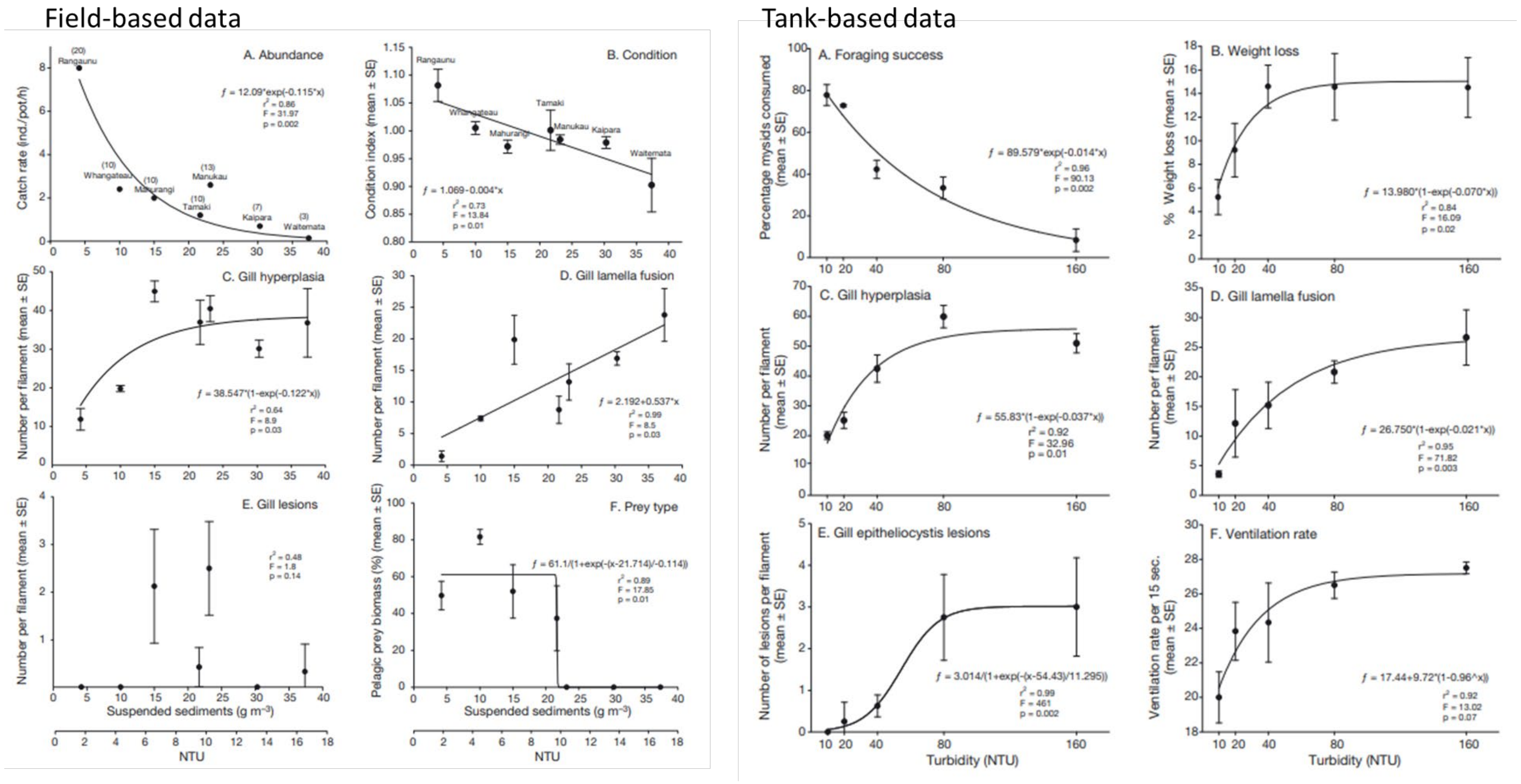


Figure 29: Left side) Field-measured relationships between the turbidity of 7 estuaries in northern New Zealand and the abundance, health, and diet of juvenile snapper *Chrysophrys auratus*. Numbers in parentheses in (A) represent sample sizes. Data are plotted against measured total suspended sediment (TSS) values, which provide higher accuracy. Nephelometric turbidity unit (NTU) values are plotted as an extra axis to facilitate comparison to experimental results, based on the strong correlation between NTU and TSS from the field data (r2 = 0.93). Right side) Tank-based effects of turbidity on juvenile snapper, in experiments lasting (A) 30 min and (B−F) 30 d. (Source: figures 2 and 4, Lowe et al. 2015.)

One postulated mechanism to explain the negative correlation between juvenile grey mullet growth and increasing proportional mangrove cover of the intertidal flats was that the mangrove forests provide relatively poor foraging areas. As the preferred ‘open’ areas for foraging grey mullet decreases through mangrove forest expansions, food limitation might occur. Mangroves themselves might not be the only driver, with the negative growth correlations with direct sediment and nutrient inputs being likely drivers. The 14 estuaries assessed were drawn from a much larger pool of 69 estuaries sampled for juvenile mullet (in part for an otolith chemistry project, Morrison et al. 2016), which collectively are thought to encompass all the major nursery areas for grey mullet in New Zealand. While ageing fish from all of these estuaries would be a large undertaking (and noting the ageing difficulties encountered by Mohd Zain 2013), future work is planned to substitute fish length at the time of sampling in lieu of age; with the expectation that juvenile cohorts moving through the population can be tracked. Preliminary data manipulations have shown promise, and, if successful, this would allow for the effects of environmental variables (included those associated with land-based human activities) on juvenile grey mullet growth to be assessed at the full scale of their nurseries for New Zealand.

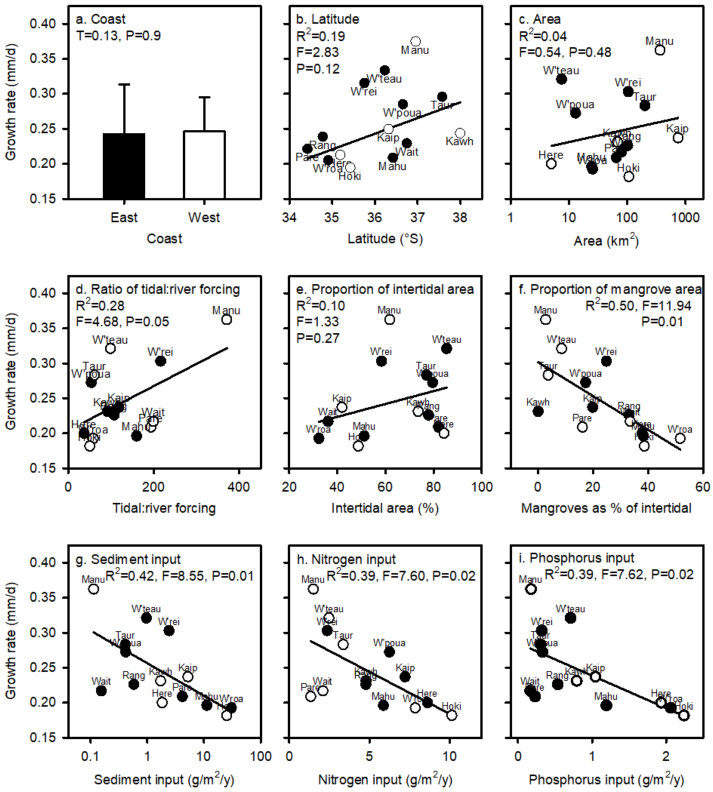


Figure 30: Correlations between juvenile grey mullet growth rate and environmental variables; coast, latitude, area, tide:river input, intertidal area, mangroves as percentage of intertidal, sediment, and nitrogen and phosphorus input. (Source: adapted from figure 9 of Mohd Zain 2013.)

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 manipulated through the addition of suspended silt/clay (< 64 microns; increasing turbidity levels of ≤ 10, 20, 40, 80, and 160 NTU). Sixty mysid shrimps (an important dietary item for snapper) were added to each tank. After thirty minutes, the fish were removed, and the numbers of surviving mysids were counted. A clear decrease in foraging success was evident with increasing suspended sediment levels, ranging from 77% in the controls (ambient seawater of ≤ 10 NTUs) to about 8% success in the tanks with NTUs of 160 (equivalent to storm conditions). Longer term tank experiments conducted over one month utilising 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 sediment 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.

6.5 Impacts on habitat formers

Indirect impacts on harvested species include the loss of habitat-forming species (Figure 31) 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 31: 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.)

6.5.1 Seagrass (*Zostera muelleri*)

New Zealand has only one species of seagrass (*Zostera muelleri*), which is indigenous to New Zealand and southern parts of Australia (previous taxonomic names have included *Z. novazelandica* and *Z. capricorni*). The conservation status of this species in New Zealand is At Risk – Declining (de Lange et al. 2017). From sampling of numerous estuaries, we know that subtidal seagrass in northern New Zealand provides an important nursery function for juvenile fish, including snapper (Figure 32), trevally, parore (*Girella tricuspidata*), garfish/piper (*Hyporhamphus ihi*), and spotties (e.g., Francis et al. 2005, Schwarz et al. 2006b, Morrison et al. 2014a–d, Morrison et. al 2019).

Seagrass meadows have been 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, Lotze et al. 2006, Pihl et al. 2006). Inglis (2003) collated and summarised data on seagrass extent in New Zealand and concluded that there had been substantial declines in many of New Zealand harbours and estuaries. Matheson et al. (2011) also concluded that there has been substantial seagrass habitat loss, particularly for subtidal seagrass habitat, within the last 50–80 years. This matches global declines in many seagrass species, which are viewed as a plant group in global crisis (Orth et al. 2006, Waycott et al. 2009). Turner & Schwarz (2006) provide a comprehensive review of what is known about seagrass in New Zealand; subsequent to that review, a range of further research has been completed, including its value in supporting fisheries species (Morrison et al. 2014a–c) and a national-scale inventory of its occurrence (Anderson et al. 2019).

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 run-off preventing adequate light reaching seagrass blades (Turner & Schwarz 2006). Large-scale losses have been documented from the Whangārei, Waitematā, Manukau, Tauranga, and Avon-Heathcote estuaries (Inglis 2003). Whangārei estuary 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). In the mid-2000s, recovery of some seagrass areas in the harbour was noticed, and this has continued to the present day. No quantitative estimate of present day coverage has been made, but informal observations suggest that around 40% of the original cover has returned. This includes areas of subtidal seagrass, which hold substantial densities of juvenile snapper and other small fish (Morrison et al. 2019). Juvenile snapper densities are still lower than those in the subtidal meadows of Pārengarenga and Rangaunu harbours to the north, but with no historical density data, it is not known whether this difference is innate, or a sign of lower habitat quality and/or larval recruitment supply (from fishing of the adult stock).

Figure 32: An extensive school (multiple thousands) of juvenile 0+ snapper (30–70 mm) passing a fixed video camera deployed over subtidal seagrass meadow in Rangaunu Harbour, East Northland. (Source: M. Lowe, MBIE CO1X0906 Coastal Conservation Management programme.)

Park (1999) used aerial photography to quantify seagrass loss in Tauranga Harbour between 1959 and 1996 and found an overall 34% decline in seagrass cover across the whole harbour over this period. The loss of subtidal seagrass was estimated at 90%. 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 based on 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, from 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 evidence for some possible recovery, as seen in places such the Tuapiro sub-estuary (Park 1999).

More recently, Ha et al. (2021) used machine-learning models on Landsat satellite imagery to assess seagrass cover (present/absent) in Tauranga Harbour. Using imagery from 1990 to 2019, they found a dominant trend of declining seagrass cover over the 29-year period (from 2237 ha to 1184 ha) (Figure 33). Seagrass was initially lost in the far northern and southern areas of the harbour during the first part of this time period, then more gradually from the central region in the second part. This study did not distinguish between areas of intertidal and subtidal seagrass.

Wide-ranging field searches in 2018 for subtidal seagrass throughout Tauranga Harbour, to catch 0+ snapper for otolith chemistry, found only one limited subtidal area with low density seagrass and, overall, few snapper were caught across four days of searching. This was despite use of Google Maps satellite imagery to pinpoint the best sites for sampling, with numerous beach seine tows made by researchers very well versed in locating higher density juvenile fish patches. Similarly, searching for subtidal seagrass in Ōhiwa Harbour found only two very limited fringing sites, which returned relatively low snapper numbers for the areas covered (M. Morrison, NIWA, pers. obs.). The habitat quality of this seagrass was poor (lower density plant cover, blades short and few per ‘sprig’), with juvenile snapper and other species known to prefer higher cover seagrass habitat.

Subtidal seagrass has been missing from Whangapoua Harbour since the early 2000s (see following section); prior to this small but healthy fringe extents were present which supported high densities of 0+ juvenile snapper and other small fish. At face value, these findings suggest that subtidal seagrass continues to be in serious decline (towards ecological extinction) in the estuaries of the Bay of Plenty and, by association, their strong underlying support of coastal fisheries production.

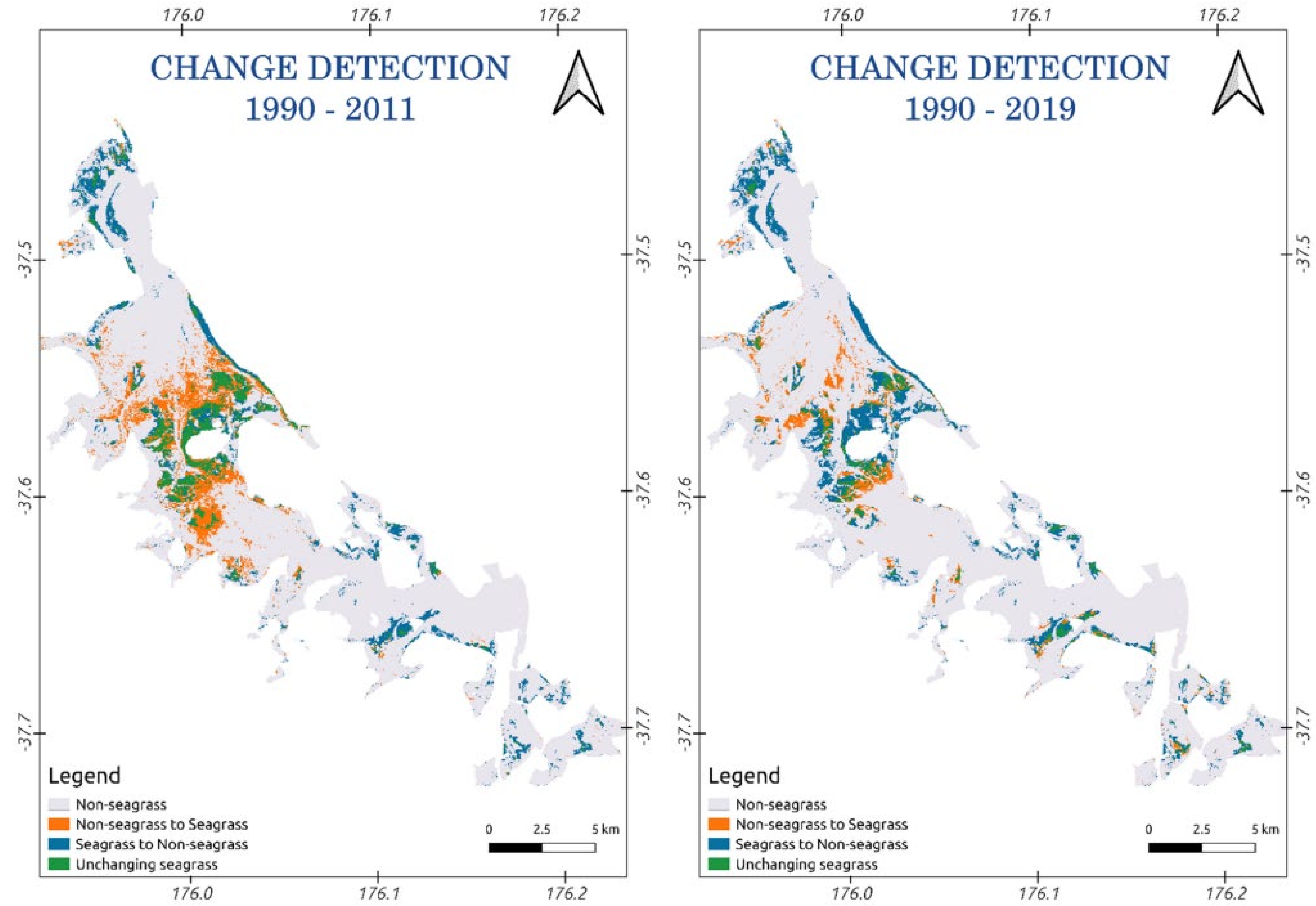


Figure 33: Seagrass change in Tauranga Harbour, from 1990–2011 (left) , and from 1990–2019 (right). (Source: figure 5 of Ha et al. 2021.)

Over on the upper North Island west coast, Morrison et al. (2014d) used Local Ecological Knowledge (LEK) interviews of long-time residents around the Kaipara Harbour to gain understanding of what historical changes had occurred. Using a relative ranking scale of change, standardised across participants, they found that most people interviewed believed that there has been a decline in intertidal and subtidal seagrass meadows over time. The scaled average ranking generated suggested a decline of around 50% between the 1930s and 1980s. Some of the older participants had memories of when fishers avoided “the grasslands”, or only set nets along the edges because of the tonnage of fish that would be caught, that at the time were considered lower value, such as parore and snapper. Descriptors such as “lush”, “vast acreages”, “50 hectares” were used, and one interviewee talked about the seagrass being so abundant that it caused bilge pumps to clog up. Some areas were identified where seagrass had disappeared or was no longer so extensive or healthy. These included the Kaipara flats off Kakanui Point near the “concrete bunker”, where intertidal seagrass was once found up into the gutters; the mid channel and off South Head; the Hōteo flats, where extensive grazing by swans was thought to have reduced the length of seagrass blades, and the entrance to the Oruawharo River, where a previously “lush” seagrass meadow of around 4 to 6 ha in size used to exist on Frenchman’s Bank off Oruawharo Heads, which was described as “one of the best” seagrass areas in the harbour. Despite this overall picture of decline, some participants indicated that there had been an increase in the extent of subtidal seagrass since the 1980s.

In 2012/2013, the habitat of southern Kaipara Harbour was mapped using aerial normal colour and infrared photography from a small plane. Processing of the infrared imagery was used to map the seagrass meadows (Figure 34). An overall seagrass area of 20.37 km^2 (2037 ha) as present, with around half of the this (around 10 km^2) being subtidal. Subsequent fish sampling of the seagrass and other mapped habitat types recorded substantial densities of 0+ juvenile snapper and other species associated with the subtidal seagrass meadows (but not the intertidal meadows, as expected).

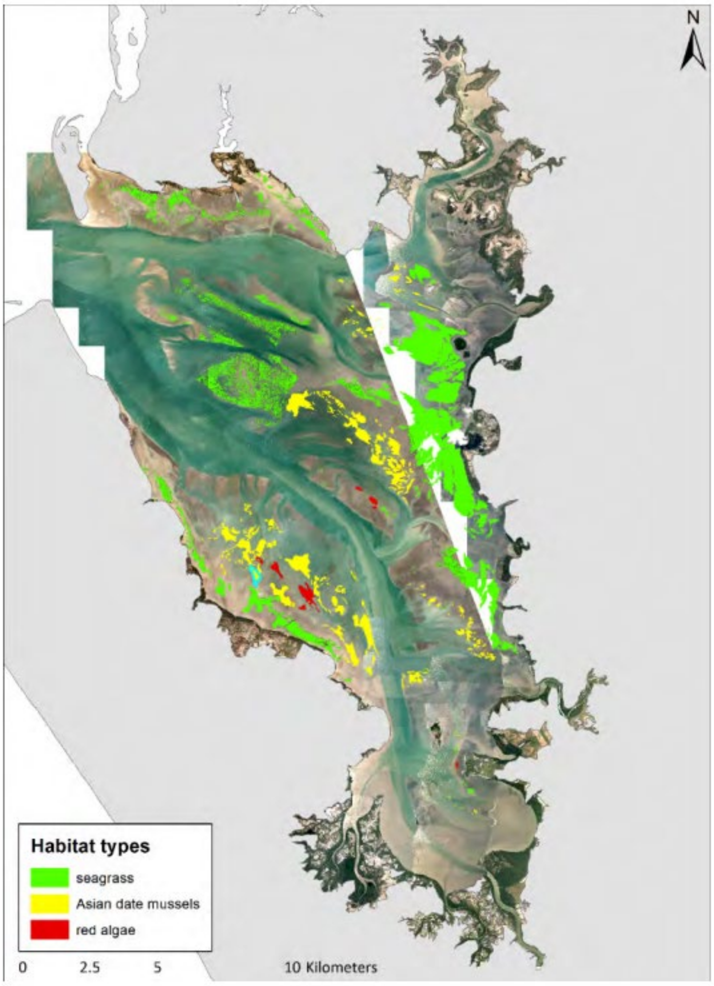


Figure 34: Seagrass and other shallow water habitats in the southern Kaipara Harbour in imagery captured on 3 May 2012 and 9 January 2013. Most of the subtidal seagrass was present in the central mid harbour area, where it predominantly occurred as numerous (many thousands) interconnected 'rosettes', with the densest areas occurring as an irregular ring in the outer part of each rosette. The 'red algae' polygons are a subset of the Asian date mussel habitat class (an invasive species; these polygons represent mussel beds with an especially heavy canopy cover (100%) of foliose red algae, but do not include all red algae growing on mussel beds. (Source figure 89 of Morrison et al. 2014d.)

Bulmer et al. (2016) looked at the factors setting the water depth limits for subtidal seagrass in the Kaipara Harbour. At the maximum depth seen for subtidal seagrass (2.10 m), light availability was measured at 2.10 (0.19 SEM) and 4.91 (0.53 SEM) mol photons m^{-2} d^{-1} during winter and summer, respectively. They found that the primary influences on light availability at the seafloor were surface light availability, the timing of low tides, and water clarity. Seagrass biomass at 2.10 m was low, indicative of limit limitation. This pattern of seagrass biomass and density 'petering' out at the lower depth end of seagrass meadows in subtidal channels in estuaries (as opposed to often sharper edges without much internal change in intertidal seagrass meadows) has also been seen in the Kaipara (and elsewhere, e.g., Rangaunu Harbour) by others (M. Morrison, pers. comm). Bulmer et al. (2016) concluded "…*suggest that the subtidal distribution of seagrass in the Kaipara Harbour is light-limited and that reductions in water clarity due to changes in land use are likely to result in significant reductions in the extent and productivity of subtidal seagrass habitat*".

Zabarte-Maeztu et al. (2021) experimentally assessed the interaction between substrate muddiness and irradiance on seagrass growth and survival over a six-week period. Seagrass was grown on two substrates from the same estuary (Pāuatahanui Inlet, Wellington): a high mud content (42%) substrate from an inner estuary site that historically held seagrass; and a moderate mud content (20%) substrate from an outer estuary site where seagrass still grew. Irradiance levels of 'low' (6.3 mol quanta m^{-2} d^{-1}) and 'very low' (2.3 mol quanta m^{-2} d^{-1}) were used (both are higher than the minimum light level required by seagrass of 1.9 mol quanta m^{-2} d^{-1}). Below-ground biomass and rhizome growth were significantly reduced by substrate muddiness but not detectably affected by irradiance. Shoot growth was reduced by both reduced irradiance and increased muddiness, with a significant interaction. The authors concluded that "*muddification of substrates imposes an increased irradiance requirement for Z. muelleri to cope with adverse rhizosphere conditions*".

Gladstone et al. (2018) undertook a multi-site manipulative field experiment in Tauranga Harbour to assess eutrophication impacts on seagrass. At six sites holding differing sediment properties and macrofaunal communities, they buried slow-release urea fertiliser (200 g N m^{-2}) in 1-m^2 plots at the start of the peak seagrass growing season (summer). After 60 days they measured seagrass morphology (cover, leaf length and width, above and below ground biomass), sediment properties, and macrofauna community structure. Resilience of seagrass meadows to N enrichment was highly site-dependent. Two sites declined in their seagrass morphology, through large reductions in seagrass cover and leaf length (up to 78%) (both important variables for juvenile fish habitat quality, Morrison et al. 2014c). The other sites appeared resilient to N enrichment. Site-specific responses were correlated with seagrass biomass and macrofaunal diversity; those with low values of these variables were less resilient to enrichment. It was suggested that the seagrass biomass level below which resilience is lost may lie between 140 and 285 g DW m^{-2} (Gladstone et al. 2018).

No updated seagrass meadow mapping at the Kaipara Harbour scale has been done since 2013. Anecdotal reports from the late 2010s suggest that subtidal seagrass had substantially disappeared from the southern Kaipara Harbour. Current Google Maps imagery (accessed 14 November 2021) shows some subtidal seagrass is present, but the imagery used is not at low tide, so most of the subtidal bank area that held subtidal seagrass in 2011/12 (Morrison et al. 2014d) is obscured by water cover. As with Tauranga, if these subtidal meadows have been lost, then the associated fisheries productivity supported by juvenile fish produced in that habitat will also have been lost.

In the outer eastern Bay of Islands, Booth (2018) assessed the temporal change of twelve subtidal seagrass meadows occurring around the island and adjacent mainland shore. Using aerial images, some extending back to the 1930s, he found that little subtidal seagrass was present before the early-1950s and suggested that the significant beds now there may be a "*recently recent biome*". A possible suggested mechanism was that the land clearances of the late 1800s may have created very damaging periods of turbid water and pulses of sedimentation, that eliminated the subtidal seagrass meadows present at that time; and that it took until the 1950s for their effects to have dissipated and for seagrass recovery to begin. Support for this suggestion come from Whangārei Harbour, where it took around 60 years for seagrass recovery to be seen following large-scale loss by (industrial) sediment inputs to the harbour in the 1960s. Booth (2020) observed an overall peak in seagrass cover in the 1960s to 1980s, followed by declines in several beds in the 1990s to 2000s, and subsequent recovery. These trajectories were "*well-correlated among the three mainland beds, and moderately well among the south-facing beds on the islands, consistent with mechanisms driving seagrass establishment and persistence operating at reasonably broad scales*". Sampling of small fish assemblages from some of these seagrass meadows in 2014 recorded high densities of 0+ snapper (Lowe et al. unpublished data, Morrison et al. 2019) (Figure 35).

Figure 35: Juvenile 0+ and 1+ snapper on a subtidal seagrass meadow, eastern Bay of Islands. The image is a frame grab from baited underwater video. (Source: M. Lowe, NIWA.)

6.5.2 Horse mussels (*Atrina novaezealandiae*)

Northern New Zealand horse mussel beds provide a nursery function for juvenile snapper and trevally, as well as supporting other small fishes such as triplefins (e.g., Morrison & Carbines 2006, Usmar 2010, Lowe 2013) (Figure 36). In southern New Zealand, they are an important juvenile nursery habitat for juvenile blue cod (Morrison et al. 2014a,b, Morrison et al. Marlborough Sounds region unpublished data). For example, 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). Subsequent work by Usmar (2010) returned year-class population estimates of 115 000 in 2006, and 321 000 in 2007 (NB: this species shows strong year-class strength variations). These juvenile snapper were associated with seafloor structure, dominated by horse mussel beds, and associated epifauna. Usmar (2010) deployed artificial horse mussel patches in Mahurangi Harbour, both with and without artificial epifauna, as well as controls, and found highest snapper numbers (at 30–50 mm) to be associated with the horse mussels with epifauna. Mean densities were about 40–120 (± 30) snapper 100 m^{-2}, compared with adjacent area densities of 4.7 (± 3) 100 m^{-2} (as sampled by beam trawling) a 10–30 fold difference, attributable to the artificial horse mussel structures. Other associated species included triplefins, juvenile spotties (*N. celidotus*), goatfish (*U. lineatus*), and the invasive bridled goby (*Arenigobius bifrenatus*) (Usmar 2010).

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 on the east coast of north New Zealand (Cummings et al. 2005). The ecology of this harbour is known to be changing because of increased sediment loading (Gibbs 2004). Early 2000s ‘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 was from a small number of storm events each year.

Figure 36: 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.)

6.5.3 Sponges

Sponges are an important component of many coastal environments, especially on rocky reefs, and on coarser soft sediments in higher current areas (Figure 37). 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). A global review of the ecological role of sponges concluded that they fulfil many important functional roles across temperate, tropical, and polar ecosystems; but that there was insufficient information to show the relative importance of these different functional roles. It was concluded that "*determining or ranking the functional roles is difficult, but it is important, since some roles that sponges fulfil will almost certainly be more important than others, and this may vary spatially or temporally*" (Bell 2008). Their role in potentially supporting fisheries was only lightly touched on. Work on the biogenic habitats on New Zealand's continental shelf in 2011 (Jones et al. 2018) found deeper water (50–250 m) rocky reef systems appeared to be impacted by sedimentation (sediment 'dustings' on rock surfaces, low biogenic species diversity and cover) in some regions (e.g., from Māhia Peninsula to East Cape). This trend was also seen on East Northland's deep rocky reefs in 2009 (LINZ OS2020 Bay of Islands programme, Morrison et al. 2010, Morrison et al. in press). These observations correlate with coastal offshore areas seen by marine geologists as having higher land-based sediment inputs (NIWA, pers. comm.). Some preliminary sediment dispersion modelling work on land-based sediment entering the Coromandel region's marine environment has also indicated that fine sediments may be transported as far offshore as the deep reefs surrounding the Alderman Islands.

Bell et al. (2015) reviewed the impact of sediment on sponges, concluding "*although the literature clearly shows that sponges are influenced by sediment in a variety of ways, most studies confer that sponges are able to tolerate, and in some cases thrive, in sedimented environments. Critical gaps exist in our understanding of the physiological responses of sponges to sediment, adaptive mechanisms, tolerance limits, and the particularly the effect of sediment on early life history stages*".

Figure 37: A high diversity mixed sponge, gorgonian, and bryozoan deep reef assemblage (around 60 m depth off Spirits Bay, Northland. While relatively close to the mainland, an absence of nearby rivers or large catchments means that the area is largely distanced from fine suspended sediment issues. (Source: TAN1105 voyage, see Jones et al. 2018.)

Pinneda et al. (2016) experimentally examined the effects of dredge-related sedimentation effects on different sponge morphologies, as a step towards understanding potential impacts on community structure and functioning (study conducted in Queensland, Australia). Ten sponge species encompassing four different morphologies (massive, erect, cup, and encrusting) were held in 1000-l tanks and exposed to a single pulse treatment of three different sediment concentrations (0, 250, 500 mg l^{-1}). They were monitored for a two-week period, and total suspended solids (TSS) and sedimentation rates (SR) were recorded throughout. TSS dropped sharply within the first two to three hours, and, with the first 48 hours, no sediment remained in suspension for any of the treatments. Most species appeared to suffer no effects, and it was concluded that a single sediment pulse less than 16 mg cm^{-1} was not detrimental to most of the sponge species studied. However, in the higher sediment treatment there was some mortality of cup shaped *Callyspongia confoederata* and small areas of tissue necrosis in other species, especially those with massive, encrusting, and wide cup morphologies (all of which are more likely to 'retain' sediments).

Recent work on a common coastal New Zealand sponge (*Crella incrustans*) used tank experiments to assess sponge function (survival, respiration, morphology) across a gradient of suspended sediment concentrations (0–832 mg l^{-1}) (Cummings et al. 2020). The treatments ran for 4 weeks. Survival was high, and oxygen consumption was not affected. An unexpected discovery was the development of apical fistules, a phenomenon never-before observed in this species. Their numbers were positively correlated with increasing SSC and were suggested to potentially be a way of raising inhalant pores to a higher position on the sponge to facilitate water pumping (or alternatively, some response to the internal build-up of sediment). Although sediments accumulated internally within the sponges, around a third had cleared all sediments from their body two weeks after the elevated SSCs were removed. It was concluded that "*the coastal environments that these sponges live in may predispose them to coping with high SSCs, and that they may also be tolerant of sediment deposition events that temporarily cover their surfaces*".

6.5.4 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 was 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) 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 cephalus*) 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 (see also juvenile grey mullet in Section 6.4).

Analysis of aerial imagery across multiple decades for some Auckland estuaries indicated expansion rates of up to 20% per year (average of 4.1%), since the late 1940s (Swales et al. 2009). Further analysis across all Auckland region estuaries suggests a rate of 3.4% per year between 1940 and 2014, and, since 1974, differential expansion rates for dwarf (23.2% p.a.) and tall mangroves (1.2%) (Suyadi et al. 2019). Modelling found that catchment forest cover, sea level, and mean annual air temperature explained about 85% of the variation in mangrove area, and that gains in mangrove forest area were correlated with periods of catchment forest clearance.

6.5.5 Kelp/seaweed forests and assemblages

Kelp/seaweed forests are a fundamental habitat element on shallow 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%, and a complete cover of sediment totally prevented attachment. Geange et al. (2014) undertook two laboratory based experiments to assess the effect of fine sediments on *Macrocystis pyrifera* (giant kelp) and *Undaria pinnatifida* spores. Spore establishment was significantly negatively affected by sediment load (40% reduction at sediment concentrations of 16 to 60 mg l^{-1}) and sediment regime (~3 times more sediment was required for 20% reduction in spore establishment when sedimentation occurred after spore settlement, rather than before). In the second experiment, sediment effects depended on sediment load: spore establishment was 2–4 times greater at 200 mg l^{-1} than at 400 mg l^{-1}; 1.36 times greater with variable rather than fixed sediment loads; and pulsed sedimentation decreased spore establishment by 59–91% (relative to a single sedimentation event).

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 of 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).

Light availability is a key driver of macroalgal assemblage presence and dynamics. Desmond et al. (2015) examined whether the availability of light explained coastal patterns in benthic macroalgal depth distribution, community composition, and standing biomass for two contrasting regions in southern New Zealand. Sites along the East Otago coast (low light, modified catchment, 2 sites) and along the northern coast of Stewart Island (high light, forested catchment, 3 sites) were selected for assessment. Each site had a similar north to northeast aspect, was subject to similar levels of wave exposure, and had a substrate of boulders, bedrock, or combination of both. The reef at each site had a similar moderate gradient sloping down to an approximate depth of 10–12 m before reaching sand. Each reef system was inhabited by a variety of sub-canopy macroalgal species with *Macrocystis pyrifera* forming the canopy. Light loggers were deployed at 2 and 10 m water depth on each reef and left to log data for 590 days (Otago sites) and 866 days (Stewart Island). Seasonal macroalgal surveys were completed using divers over the four seasons (December 2012–October 2013). The modified coast had a consistently higher light attenuation than the forested coast at both depths, with 14 days where light was undetectable at 10 m (including a continuous 8 day period, 17–24 June). No no-light days were detected for the forested coast. At 2 m depth, 4 no-light days were recorded on the modified coast (there were none for the forested coast). The surface light environment of the modified and forested coasts was similar throughout, with average daily doses of 14.0 ± 0.42 and 13.6 ± 0.33 mol photons m^{-2} d^{-1}, respectively.

In the 10-m depth strata, the forested coast had significantly greater standing biomass than the modified coast for all four seasons. The same trend was seen in the 2-m depth strata (aside from no significant difference in spring). These differences scaled to around 3–5 and 2–4 times greater standing biomass for the forested coast (at 2 m and 10 m, respectively). Individual macroalgae on the forested coast were on average three times the biomass of individuals from the modified coast. For macroalgal species shared by both coasts, their average depth range spanned 6.9 ± 1.63 m on the forested coast and 4.7 ± 0.65 m on the modified coast. The depth limit of the shared species was on average 0.9 ± 1.08 m deeper within the forested coast compared with the modified coast (noting that some species may have extended to deeper depths if the reefs had gone deeper than 12 m). Of the dominant species recorded during depth distribution analysis, the two coasts shared 18 species; 7 species within the modified and 6 within the forested coast were not shared.

Tait (2019) quantified the spatial and temporal variation in photosynthetically active radiation (PAR) in forests of the giant kelp *Macrocystis pyrifera*, across a gradient of exposure to land-derived sediments (the North Otago coast, with 8 sites across 40 km of coastline). Exposure to higher suspended sediment levels pushed PAR below critical thresholds for net sporophyte growth (< 1.0 mol m^{-2} d^{-1}), and negatively affected giant kelp densities. Negative impacts were also quantified for a range of wider assemblage metrics including fucoid density, laminarian density, beta diversity, subcanopy composition, and urchin density. While kelps and fucoids (two different taxonomic groups of macroalgae) responded to similar light thresholds, kelps appeared to be vulnerable to physical disruption by sediments, whereas fucoids were tolerant of physical disruption, but vulnerable to declining light availability. Light thresholds of 1.0 mol m^{-2} d^{-1} were suggested as a useful target for the management of stressors affecting water clarity (e.g., sedimentation, eutrophication), across diverse groups of aquatic photoautotrophs (plants).

Such macroalgal species, including kelp forest and macroalgal 'gardens', 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 38). 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 38: 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 known.

6.5.6 Other potentially important habitats

There are other potentially important habitats, about which we know very little, both in terms of what harvested species may use them at some stage during their life cycle, and of the threats land-based activities pose to those habitats. One of these is maerl beds (also known as rhodolith beds) (Figure 39), 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) found a high diversity of associated invertebrate species, along with the presence of 0+ blue cod (W. Nelson, NIWA, Wellington, pers. comm.). Extensive fish-habitat sampling across East Northland including some sites over rhodolith beds suggests that where epifauna such as the green fleshy macroalgae *Codium* are growing attached to rhodolith beds, juvenile fish including snapper, spotties, and leatherjackets may be found in close association. However, the rhodolith individuals themselves, absent of attached fauna, do not show such associations. Towed video observations of rhodolith beds along the open coastline of the Marlborough Sounds also do not show any associations of juvenile blue cod with these habitats.

Figure 39: Example of a rhodolith/maerl bed. The pink and red colours are living plant tissue.

Large beds have been documented in Northland off the Cavalli Islands (Grace & Hayward 1980), off Urupukapuka Island, Bay of Islands (Hayward et al. 1981), and on the southern side of Little Barrier Island. 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 years (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. 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 y^{-1}) substantial deposits take centuries 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 40) 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), and skate (*Raja nasuta*) and shark egg cases were commonly found attached to the reefs.

Figure 40: 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. 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 terrigenous 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 macroalgae species that lives on soft sediments, were also mentioned.

In 2019, fields of *Galeolaria* biogenic patch reef were discovered in the eastern Hauraki Gulf (Figure 41), expanding their known New Zealand distribution 500 km north (previously only known as far north as the Marlborough Sounds, Morrison et al. 2014). Multiple small fields were revealed by multibeam sonar around most of Pakatoa Island on the west side of the Firth of Thames, as well as to either side of Motumorirau Island (north of Coromandel Harbour, west Coromandel Peninsula), totalling around 1 km^2 in extent. LINZ multibeam sonar data from 2021 off Coromandel Harbour, including around the Cow and Calf islands, and Rangipukea Island, has since revealed additional new (albeit small) likely fields. More tubeworm fields are likely to exist around the islands that form the Motukawao Islands/Happy Jack Islands, just to the north, and around Deadman's Point, just to the south. A healthy and epifaunally diverse *Galeolaria* field was observed by divers off Moturua Island in 2021, one of the Happy Jack Islands (Shane Kelly, Coast & Catchment Ltd., pers. comm.; Shaun Lee, independent blogger, pers. comm.). This biogenic reef forming species may have been historically much more widespread in the gulf in mound field form. Its resilience to increased sedimentation is unknown. The Pakatoa fields visually look to be healthier and with less associated fine sediment that those around Motumorirau Island. Towed cameras over some of the mounds have revealed a rich epifaunal assemblage (e.g., sponges, ascidians, bryozoans) associated with the Pakatoa Island fields, whereas those on around Motumorirau Island have less associated epifauna and appear less biodiverse. Two beam trawl tows in areas adjacent to these fields inadvertently caught *Galeolaria* mounds: one in 2017 off south Rotorua Island and one in 2020 off Hautapu Rocks, west Coromandel. Both mounds held numbers of bastard red cod, rockfish (*Acanthoclinus* sp.), and black feathery brittle stars. Limited baited underwater video and trap drops (2020) have also shown snapper associations with these mound fields.

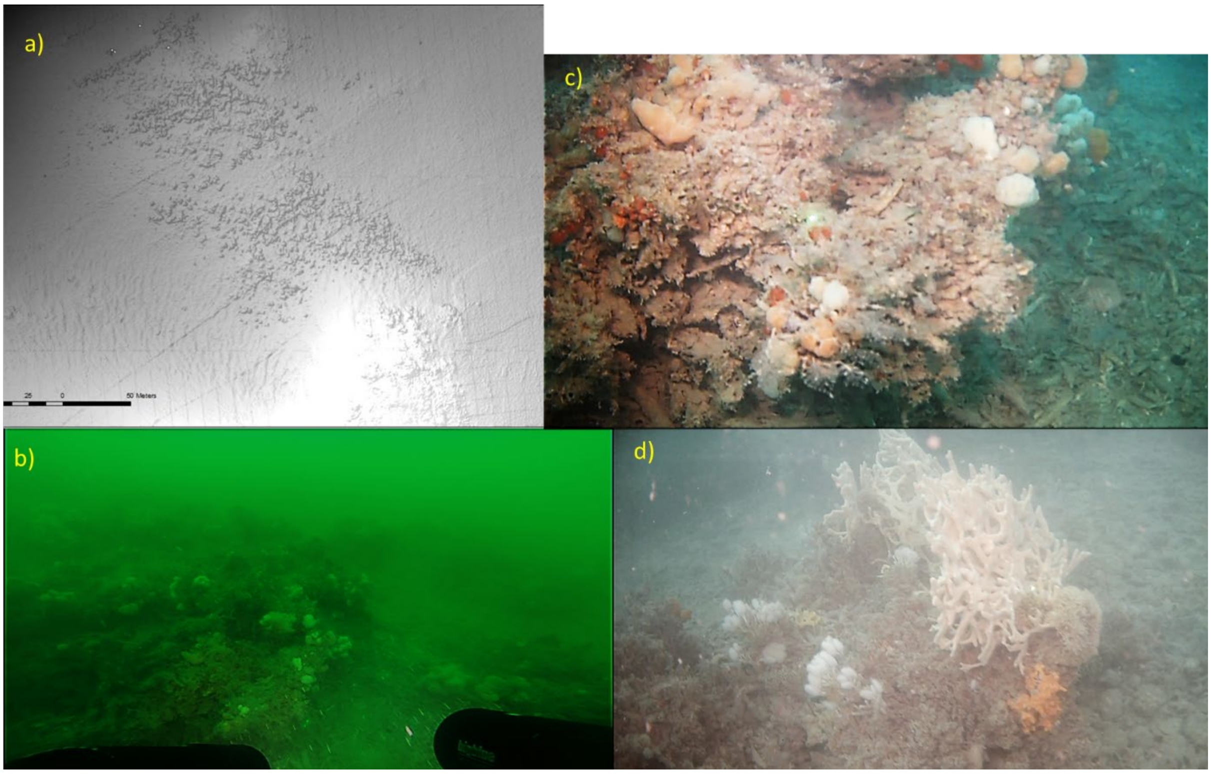


Figure 41: ***G. hystrix*** **mound fields in the Hauraki Gulf. A) multibeam sonar bathymetry of a field just north of Pakatoa Island, b) a video landscape view of the same field, c and d) closer views of the attached epifauna associated with the mounds. (Source: MBIE Juvenile fish habitat bottlenecks programme (C01X1618) imagery, 2019–2020.)**

6.6 Pits and burrows in soft-sediment habitats and land-based impact on them

6.6.1 Why pits and burrows are important

‘Ecosystem engineering’ is a term coined to describe the construction, modification, and maintenance of biologically generated habitat features by organisms (Jones et al. 1994). In marine soft-sediment environments, most examples of ecosystem engineering link directly or indirectly to bioturbation and bio-irrigation. Bioturbation is the process by which burrowing organisms create and modify sedimentary environments by mixing and moving sediment particles, for example, when excavating burrows or deposit feeding. Bio-irrigation refers to the flushing or ventilation of sediments with water for the purposes of filter feeding, removing waste, and accessing dissolved oxygen.

Both bioturbation and bio-irrigation, and thus the activities of animals that move particles and fluids in taxon-specific ways, are fundamental to our understanding of bioengineering and habitat modification in soft-sediment marine habitats. These activities have multiple interrelated consequences for marine seafloor habitats influencing processes such as sediment transport, carbon trapping, primary production and nutrient recycling (Lohrer et al. 2004, Mermillod-Blondin & Rosenberg 2006, Thrush et al. 2006, Le Hir et al. 2007, Lohrer et al. 2010), ultimately altering the ways in which ecosystems function. The magnitude and direction of effects generally relate to the physical and behavioural attributes of the organisms involved (e.g., large body size, depth of burrowing or body position within the sediment, burrow complexity or frequency) and/or their population densities.

The modification of microtopography through the creation of pits and mounds influences the hydrodynamics of benthic boundary layer flows and creates spatial variation in hydraulic pressures that can move dissolved solutes (oxygen, nutrients, metabolic wastes) in and out of the sediment (Zeibis et al. 1996, Huettel et al. 2003). Burrows also create a heterogeneous three-dimensional matrix within the sediment column, extending the sediment-water interface. Burrow networks have been likened to veins and arteries in marine sediment systems because they act as conduits to transport oxygen, metabolites, particles, and nutrients between reaction zones (Reise 2002). They also provide extensive habitat for microbial communities that exert strong control over biogeochemical processes. Burrow ventilation, which can occur both passively and actively, is closely coupled to bioturbation, because the latter subducts labile material whilst the former speeds up reaction rates through increased oxygenation and exchange (D'Andrea & DeWitt 2009, Pillay et al. 2011, Pillay 2019 and references therein). These processes alter the pathways and magnitude of solute fluxes and enhance benthic-pelagic coupling (Nielsen et al. 2003). The activities of bioturbators and bio-irrigators can have spatially widespread influence (e.g., at the scale of entire coastal systems, Seike et al. 2020).

Notable bioturbators and bio-irrigators include crabs and several lineages of burrowing shrimps. These species can create holes, pits, and mounds in sediment systems. Many bivalves and polychaetes create holes, pits, and mounds also. Although densities of burrow builders are generally greatest in intertidal and shallow subtidal habitats, there are species that create burrows in continental shelf sediments down to depths > 1500 m (e.g., scampi and stomatopod shrimps).

Bioturbators rework sediments using a suite of burrowing techniques (Atkinson & Eastman 2015) to create temporary or permanent burrow refuges of different sizes, shapes, and depths associated with their lifestyle traits. Through the physical process of burrowing, sediment is mixed between different depth horizons and reaction zones. Burrowing mixes both particles and porewaters and can have considerable effects on sediment transport, water column turbidity, nutrient dynamics, and benthic community composition, all of which can have system level consequence for functioning. Burrow construction modifies the sedimentary milieu, altering grain size distribution, sediment permeability, bulk density, and shear strength (Grabowski 2011), ultimately influencing the geomorphic evolution of systems, particularly in shallow coastal zones (Crotty et al. 2020). Many burrowing species have been shown to increase sediment water content and penetrability due to the loosening of the sediment fabric whilst building or maintaining burrows. This in turn can stimulate microbial activity and organic matter processing (Watling 1998) and influence the rates and pathways of sediment diagenesis.

Thalassinidean shrimp have some of the highest rates of sediment processing of all benthic organisms (Branch & Pringle 1987, Witbaard & Duineveld 1989, Rowden et al. 1998, Pillay 2019) and form some of the deepest and most complex burrow structures (Dworschak 1983). Inhalant and exhalant openings connect burrow interiors with overlying water. Burrows are continuously maintained by the resident shrimps, who eject processed sediment and faecal deposits to the sediment surface. This is how volcano-like mounds form and create topographic relief on the surface of the sediment. Rowden & Jones (1994) estimated that faecal deposits increased the roughness height of the seabed by more than 1000-fold in the presence of high densities of *Callianassa yessoesnsis*, inducing turbulent benthic boundary layer flows.

Unlike the filter feeding thalassinidean shrimp, deposit-feeding species (particularly mobile crab species) often modify the sediment surface by building burrows with a broad and deep aperture. These high aspect ratio openings have been shown to serve as sediment traps, capturing fine particles and organic material at depth and altering the sediment granulometry over time (Yager et al. 1993, Le Minor et al. 2020). Subsurface burrow shape can also increase passive sediment retention, creating a highly effective and energy-efficient means of food capture and storage (Escapa et al. 2008). Material from the bottom of the burrow can be brought back to the sediment surface during burrow maintenance or integrated into the sediment matrix at depth.

The importance of burrowing species to nutrient cycling is well documented (Forster & Graf 1995, Nickell & Atkinson 1995, Webb & Eyre 2004, Mermillod-Blondin & Rosenberg 2006, D'Andrea & DeWitt 2009, Dunn et al. 2009, Needham et al. 2011). The temporal dynamics of solute exchanges within and around burrow structures is linked to labile carbon content, organism activity rates, and behaviour, because the movements of animals alter porewater pressure gradients, particularly in advective sediments (Laverock et al. 2011, Volkenborn 2012). At an ecosystem scale, active irrigators can pass significant volumes of the overlying water through their burrow structures resulting in broad-scale changes to nutrient dynamics. Burrowing enhances particulate organic matter capture and transformation into inorganic nitrogen, a key process for fueling primary production.

6.6.2 Why pits and burrows are important to fish

Habitat modification by large or abundant burrowers is known to impact benthic community composition (Brenchly 1981, 1982, Posey 1986, Posey et al. 1991, Berkenbusch & Rowden 2000, Widdicombe et al. 2000, Sumida et al. 2020). One element of this is the use of burrows by a range of sedentary and mobile species, from tubiculous polychaetes to gobiid fishes. The burrows appear to provide effective refuges from predation (e.g., from demersal fish). Gallery-style burrow structures, with multiple surface entrances/exits, are probably particularly useful as refuges (i.e., those hiding are not forced to leave the same hole they were chased into, reducing predation risk).

Fish do not have to enter burrows to benefit from these features. That is, the topography provided by pits and mounds can have value as refuge habitat for fishes. As an example, surveying to identify and map the 0+ snapper nurseries of East Northland and the greater Hauraki Gulf has revealed two different groups of preferred seafloor habitats. One is emergent 'upward facing' biogenic habitat structure (e.g., subtidal seagrass meadow and horse mussel beds). The other is mud seafloors that are heavily pocked with larger burrow complexes, i.e., gallery-style burrow structures, probably created by crab species (i.e., 'downward facing' biogenically created structure, as opposed to flat mud seafloors with no large burrows) (Figure 42). The 0+ snapper (10–90 mm) are mainly zooplankton feeders, and it is quite likely that these burrow complexes provide some form of shelter when needed from large predators (e.g., kahawai, larger adult snapper (cannibalistic), John dory, kingfish). This juvenile snapper and mud with burrows association is seen across three of the four main nursery grounds of the Hauraki Gulf, in the eastern Bay of Islands, and smaller estuaries such as Whangaroa and Whangaruru (upper east Northland).

Figure 42: Burrow complex on soft muddy sediment from a juvenile snapper nursery in the Hauraki Gulf. This image was taken using a towed fibre optic camera; fine flocculent material can also be seen on the seafloor surface. (Source: MBIE CO1X1618 'Juvenile fish habitat bottlenecks' programme.)

6.6.3 How land-based effects may influence pits and burrows and soft-sediment habitat relief

The effects of terrigenous sediment on the abundance of pits and mounds is likely to be taxon specific. Mud crabs that can burrow through the deposits have been observed to survive major deposition events, unlike many of the less capable burrowers (e.g., bivalves, A. Lohrer, NIWA, pers. Obs.). The New Zealand crab *Austrohelice crassa*, which builds shallow burrows in the intertidal zone of estuaries (Needham et al. 2011), has been shown to be an early coloniser of terrigenous sediment deposits (Norkko et al. 2001). Its burrowing tends to break up the surficial clay layers and enhance the erosion of the deposits, facilitating overall community recovery. Even in relatively thick deposits (up to 10 cm thickness), large specimens of *A. crassa* (2–3 cm carapace width) have been shown to penetrate the clay layer and mix the fine material into the sediment matrix at depth (Gibbs et al. 2001). Thus, the ability of bioturbating and bio-irrigating species to turn over the upper part of the sediment column has implications for disturbance and recovery from land-based impacts. However, sediment loading—unquestionably bad from a biodiversity and ecosystem function standpoint—may only have minor impacts on the presence and abundance of pits and mounds in soft-sediment systems.

As with sediments, excessive loadings of nutrients and organic matter from land to sea can impact estuarine and coastal systems. However, the effects of nutrients and organic matter on burrowing species are likely to be indirect. That is, nutrients and organic loadings have the potential to influence sediment biogeochemistry and bottom water oxygen concentrations, which in turn could affect the suitability of the sedimentary environment for pit- and mound-forming burrowing species. The link between terrigenous nutrients and bottom water hypoxia is well understood. Pit- and mound-forming macrofauna tend to be relatively sedentary and cannot avoid (move away from) hypoxic or anoxic conditions. Therefore, repeated or protracted hypoxic/anoxic events are expected to negatively impact these animals. Oxygen depletion events are known to be seasonal and dependent on water column temperature, productivity, and stratification (Stenton-Dozey 2013).

Although bottom water oxygen depletion is a major concern overseas (e.g., Gulf of Mexico and Baltic Sea dead zones), the only area in New Zealand that appears to have recurring issues of bottom water hypoxia at present is the Firth of Thames (Zeldis et al. 2019). In the Firth, however, there have been 166 recorded events of < 70% dissolved oxygen saturation over the past 15 years, 7 of which lasted > 20 days (the longest being 66 days, Zeldis et al. 2019). Dissolved oxygen concentrations of < 70% saturation already occur 56.7% of the time in this area during summer. Climate-related warming in combination with further nutrient and organic matter inputs will likely exacerbate the problem.

6.6.4 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.

7. NEW ZEALAND CASE STUDIES

7.1 Place-based studies

7.1.1 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. Jones (2008) described the land use in the catchment as 54% exotic forest, 20% native forest and scrub, and 17% pasture. The native forest and scrub are 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 2002). 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 more than doubled, and seagrass cover more than halved, from the 1950s to the early 2000s (Jones 2008). Subtidal seagrass meadows 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). 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 43). 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 land 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).

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).

Fish-habitat survey work in the Whangapoua Harbour in 2003 found small subtidal seagrass fringes at the edge of several of the large intertidal meadows around the harbour, which supported relatively high densities of juvenile snapper, parore, spotties, and other small fish. By 2007, when a return trip was made to run a habitat-mediated snapper survivorship experiment, no subtidal seagrass remained aside from two very small patches (< 5 m^2 each) downstream of an oyster farm. The experiment was abandoned due to the harbour-wide absence of the small juvenile snapper required. Since that time, regular checks on Google Maps have not revealed any return of subtidal seagrass anywhere in the harbour (as of November 2021, a 14-year period).

Figure 43: 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.)

7.1.2 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, which covered some 55 km^2 of seafloor in 2003 (Figure 44) (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 45). As these beds were impacted and declined, so did the proportions of juvenile fish (snapper and tarakihi) in the catch (Saxton 1980a). At Separation Point, the more robust species *Celleporaria agglutinans* dominated, causing fishers to avoid the area up 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).

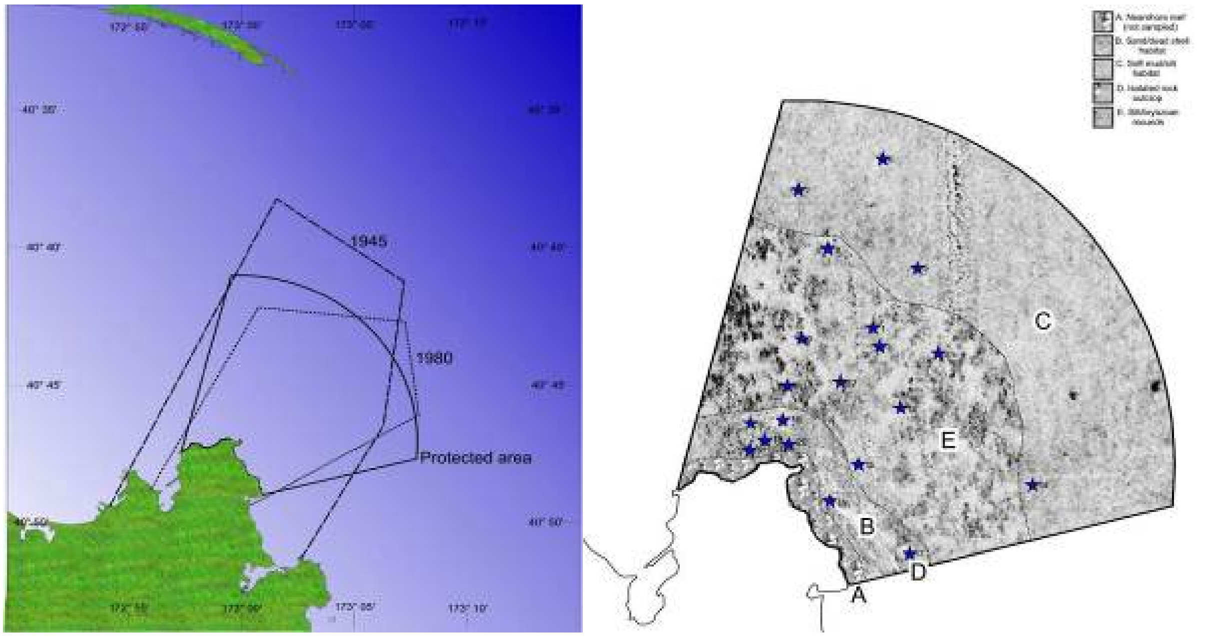


Figure 44: Left: Location of Separation Point bryozoan bed protected area, and extent of the beds themselves as in 1945 and 1980, based on fisher observations. Right: digital side-scan imagery mosaic, showing the present day bryozoan bed extent (habitat ‘E’). (Source: Ken Grange, NIWA.)

Unfortunately, significant damage to the bryozoans occurred following the introduction of pair trawling, 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 up 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, much 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 bryozoan 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 area 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 45). 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 can detect 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 more 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 occurred following human-driven forest clearance. It was also speculated 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). Modelling different river flows, wind directions, and velocities, it was shown that the Motueka River influence could extend into the western side of Tasman Bay and, during flood conditions, 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 consistent with existing bathymetric and seabed substrate characteristics. Model runs with 20 knot winds in either a north-easterly or south-easterly 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.

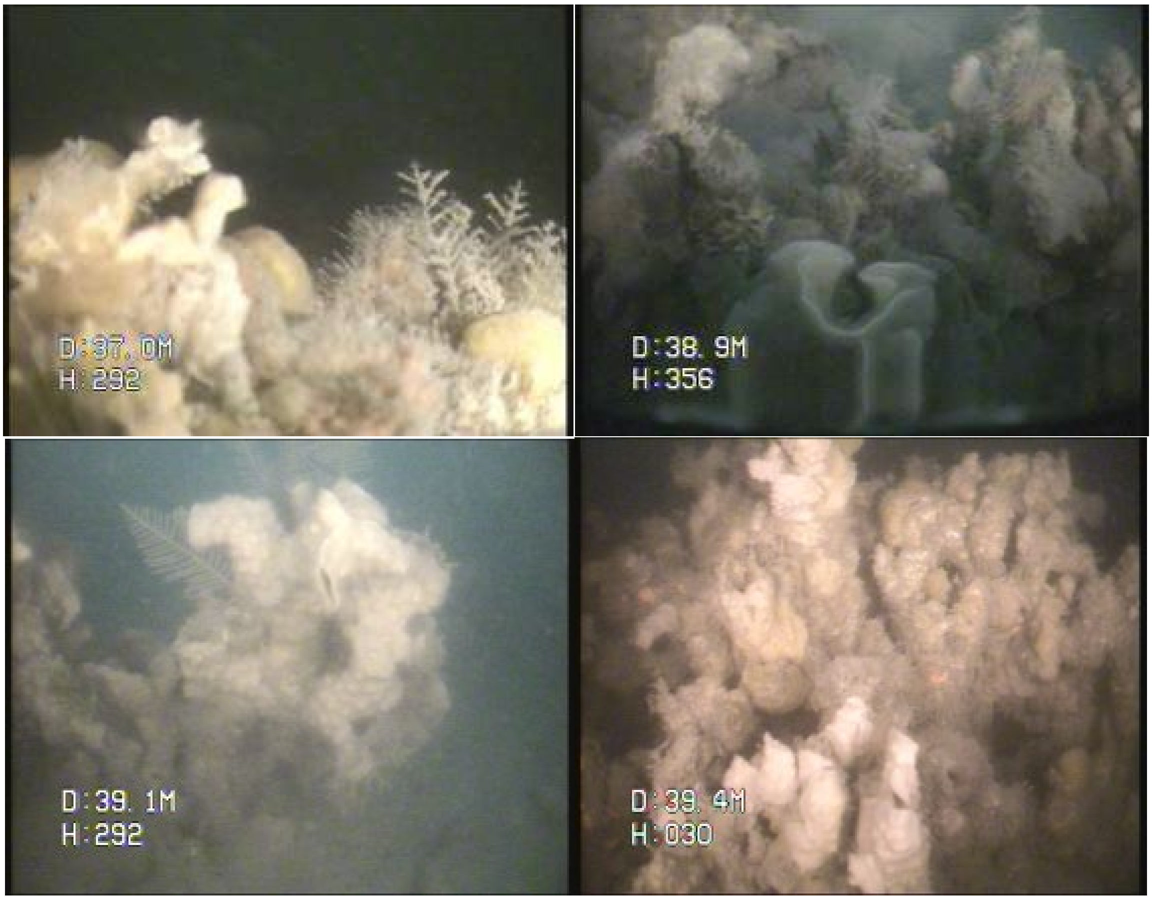


Figure 45: Bryozoan mounds at Separation Point in 2002, including a) hydroids and sponges, b) bryozoan (*Hippomenella*) and encrusting fauna, c) bryozoan (*Celleporaria*) and hydroids, d) bryozoan colony (*Celleporaria*). (Source: Ken Grange, NIWA.)

In late 2020, the closed Separation Point area and a 1-km wide buffer on its western and eastern sides was fully mapped by multibeam sonar as part of the MBIE funded ‘Juvenile fish habitat bottlenecks programme’. The resulting high-resolution seafloor map showed a wide range of features, including hundreds of harder material patches/rings likely to be bryozoan mounds. In February 2021, this map was used to stratify the area by putative seafloor habitat type, and a stratified-random survey was deployed using towed camera systems and baited fine mesh fish traps with Go-Pro cameras. Sediment samples were also collected at each station using a grab.

Very few bryozoan colonies (or any other biogenic habitat-forming species) were observed, and the few species seen in low numbers were dominated by ascidian species that appear resilient to high sedimentation levels (also seen in other sediment-stressed areas, e.g., Pelorus Sound). Many large patches of dead carbonate material (shells, bryozoan fragment, and other debris) were present (Figure 46), matching the positions and spatial scale of the harder seafloor patches mapped with multibeam. These patch mosaics were surrounded by soft featureless muds which formed the dominant habitat. Close to shore, a few low-density horse mussel areas were located, as well as zones of 100% cover of dead turret/screw shells. Few fish were seen on the towed videos, with no juvenile blue cod present. Similarly, the baited traps had very low catch rates, mainly of larger, older blue cod (catch rates of < 0.5 fish per pot), with no juvenile snapper or tarakihi being caught (or seen with Go-Pros). The underwater visibility was poor, although there was calm settled weather and better visibility than normal for the area (e.g., towed DTIS camera deployments in 2011 encountered very turbid water, and little of the seafloor was visible, Jones et al. 2018). Essentially, the extensive bryozoan fields reported for this closed area are now gone, leaving behind only extensive patches of carbonate rubble. While data analysis and associated interpretation are only now commencing, the most likely cause is Cyclone Gita (in combination with land-use change), which swept directly across the area in 2018 with great force and caused widespread destruction. Grain size analysis has returned a strong dominance of muds and silts for all sites sampled, including six sites that also have coarser ‘gravel’ material present (likely to be dead bryozoan colonies carbonate debris).

Storms similar in size to Cyclone Gita are relatively frequent in New Zealand (Griffiths et al. 2020). Between 1970 and 1997, tropical cyclones arrived in New Zealand at a rate of slightly over one a year (Sinclair 2002). For the Separation Point region (Figure 47), storms with a similar precipitation (rain) intensity as that of Cyclone Gita (>130 mm in 24 hours), occur every 1 to 5 years (Griffiths et al. 2020; from Tasman District Council; recurrence estimates drawn from 1962 to 2019 from Takaka Hill, Riwaka South, and Motueka at Woodmans rain gauges).

Figure 46: Left) seafloor image from inside the closed Separation Point area, showing carbonate debris and ascidians (pale yellow), at a site that previously supported bryozoan colonies (lasers are 10 cm apart); right: a baited fish trap set on soft mud also inside the closed area, with two adult tarakihi and a 35 cm snapper (inside the trap).

Figure 47: Examples of suspended sediment plumes from western Tasman Bay extending around Separation Point. (Source: Griffiths & Robinson, PowerPoint presentation to Tasman District Council 2021.)

Griffiths et al. (2020) used gridded rainfall, topography, lithology, and land cover surfaces to develop a high-resolution (10 × 10 m) spatial model of landslides that occurred in this region during the period that ex-tropical Cyclone Gita bought heavy rainfall. They also separately modelled landslides in the same dataset, as a function of the erosion susceptibility classification (ESC) data layer used to manage forestry activities (under the National Environmental Standards for Plantation Forestry, NES-PF). Models were fitted using boosted regression trees. Their preferred model had very high predictive power (AUROC = 0.93) and included parameters for aspect, elevation, mid-slope position, land cover, rainfall, slope, and a descriptive seven-class topographical index. Land cover, elevation, rainfall, slope, and aspect were the strongest landslide predictors: with the land cover classes of seral native vegetation and clear-felled plantation forest predicting higher probabilities of landslides; and the land cover classes of tall native forest, and closed canopy plantation forest, predicting lower landslide probabilities. The ESC was a poor predictor of landslides in the study area (AUROC = 0.65). This showed that accurate, high-resolution landslide probability surfaces can be developed from landslide distribution, land cover, topography, and rainfall data. They also concluded that "*landslide occurrence in the Tasman region could be substantially reduced by increasing the extent of permanent forest cover and by limiting clear-fell harvest of plantation forests on landslide-prone slopes*".

The loss of the Separation Point bryozoan fields 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. Whether these systems will recover is unknown. The findings of Griffiths et al. (2020) provide a clear example of how land-based activities create higher sedimentation levels in this region's coastal waters, and also provide a clear signpost to what can be done on land to help recover and improve the health and productivity of the regions coastal fisheries. As with the example of Cyclone Bola on the lower east North Island, such weather events are not uncommon, and will increase in both their intensity and frequency as climate change progresses.

7.2 Species-based studies

7.2.1 Habitat connectivity

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 (Note: as discussed above, as of 2021 the Separation Point bryozoan beds are non-functional).

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.

7.2.2 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 economically unviable due to very low population densities. Before this, a large reserve of many age classes would have 'buffered' the stock for several decades. 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 in the seven main west coast estuaries (Whangape–Kāwhia), 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.

Sampling inside the seven estuaries in 2003 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 48) 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.

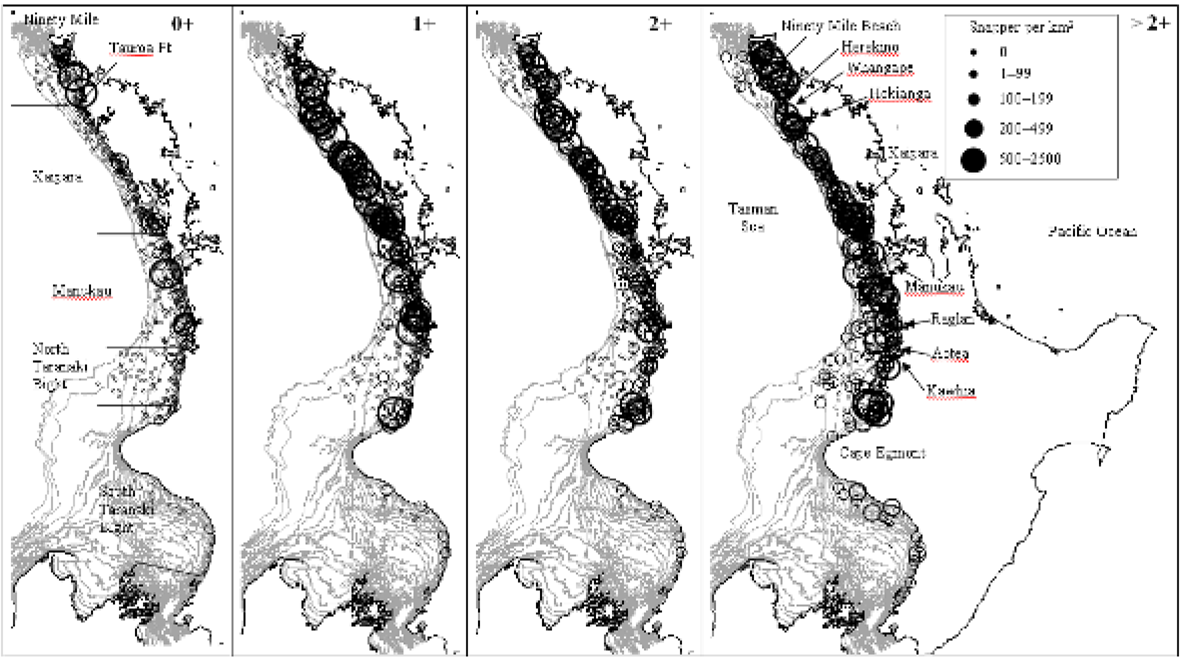


Figure 48: Snapper densities per km^2, by age class (0+, 1+, 2+, and >2+), along the west coast of the North Island, as sampled by Ministry of Fisheries trawl surveys (1982–1999). Positions of estuaries sampled are shown (except Herekino estuary). Also shown are the five coastal sampling zones from which 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 (see Seagrass Section 6.5.1). Subtidal seagrass meadows are not known to occur in any of the other estuaries sampled. 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.

As discussed elsewhere, many of the complex three-dimensional habitats in which juvenile snapper were found in association with such as horse mussels, seagrass, and sponges 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 contributes relatively little to the coastal population based on otolith chemistry. 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 (Morrison et al. 2014d), 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. unpublished results).

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

Estuaries and sheltered embayments are now well accepted as important fish nurseries in both tropical and temperate systems globally. For most of the east coast of the South Island, only estuaries offer sheltered waters and may be disproportionately important as nurseries for some fish species along this coast. Work by Colman (1978) tagging juvenile sand flounder demonstrated such functions for three Christchurch region estuaries (Avon-Heathcote, Lyttelton, and Akaroa). We know that these estuaries have undergone change over time, not least from the 2011 Christchurch earthquake which caused large changes to the Avon-Heathcote harbour and catchment, including many thousands of small mud liquification 'volcanos' on the harbour tidal flats. We have little understanding of the present day status of these estuaries as juvenile sand flounder nurseries, more than 50 years after the tagging study was completed.

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 49) (Colman 1978). Recaptures were recorded from recreational and commercial fishers, as well as some by researchers: 3307 tagged fish were returned with sufficient associated location data to be used to assess movement.

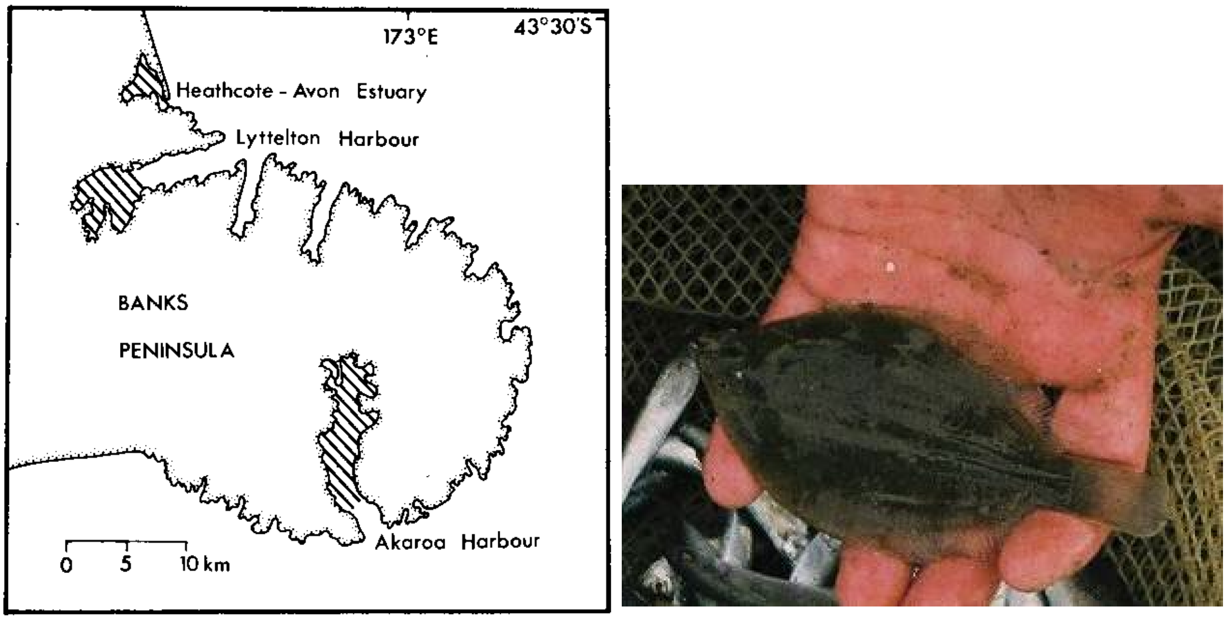
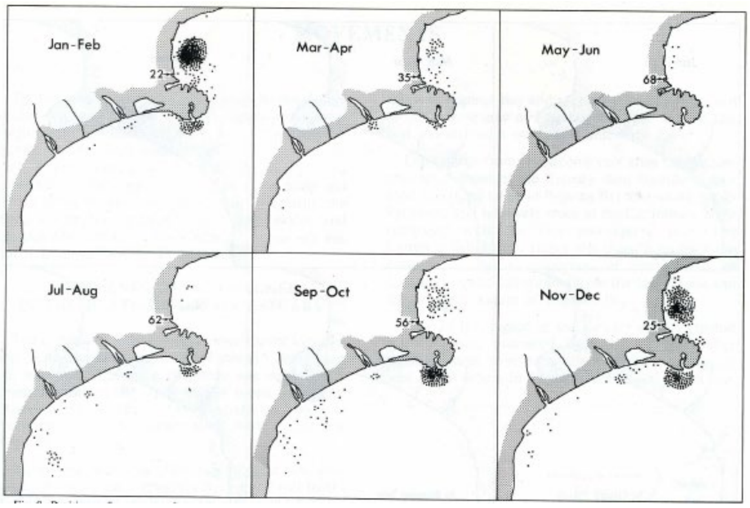
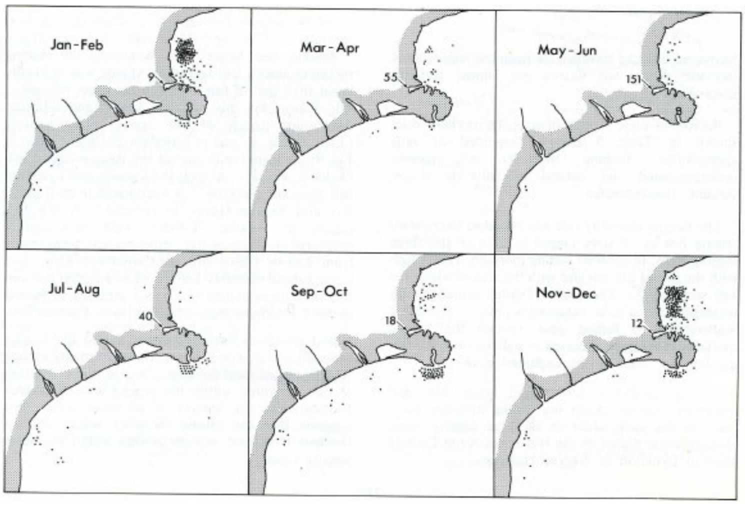


Figure 49: (left) Estuaries in which sand flounders were tagged – Avon-Heathcote (5767 fish), Lyttelton (3 390 fish), and Akaroa (5894 fish); (right) Juvenile sand flounder, sampled in Kaipara Harbour. (Sources: a) Colman 1978, b) Crispin Middleton, NIWA.)

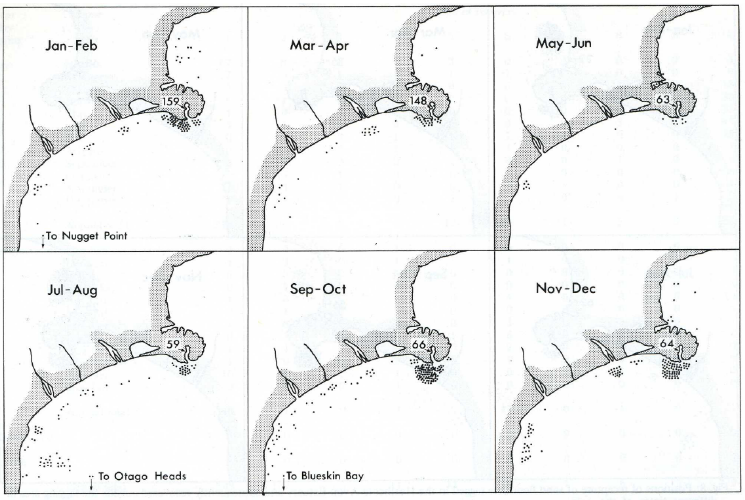
Most flounders left the estuarine tagging areas within a year of being tagged. Increasing returns from coastal areas, in particular Pegasus Bay and the waters off Akaroa Head, occurred from August to November, and this was taken to be when most of the flounder moved out of the estuaries. This was at 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 50). 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 another 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 50: 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 (and yellow-belly) 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. One-off sampling of the three Banks Peninsula harbours in 2005 found relatively low numbers of sand flounders (Francis & Morrison, unpublished data).

8. IWI AND HAPŪ PERSPECTIVES

In the following section we have collated accessible information on the effects of land-based stressors on kaimoana, as expressed by iwi and hapū around Aotearoa New Zealand. The key purpose is to provide background into some of the concerns and interests held by selected iwi and hapū around land-based stressors and their impacts on kaimoana, particularly due to sediment. For those unfamiliar with Te Reo Māori words and phrases, a glossary is included at the back of this report (Section 12). We encourage readers to source the cited references for a more in-depth understanding of Māori cosmology, principles, and concepts and the tribal and rohe histories touched on in this section.

8.1 Methods

Perspectives on the issues covered by this review will vary between iwi, hapū, whanau, and marae, which have been developed over time through their interactions with their freshwater and marine environments. To demonstrate this, we accessed selected Iwi Environmental Management Plans (IEMPs), Waitangi Tribunal reports, and iwi/hapū-driven research and reports to provide some examples of the impacts of land-based stressors on kaimoana and associated cultural values, uses, and practices, as expressed by various whānau, hapū, and iwi around Aotearoa New Zealand. No new data collection (e.g., interviews) or analyses were undertaken to inform this study.

This section does not intend to represent the viewpoints of all iwi/hapū. While similarities exist between different communities, assumptions should not be made that all values and perspectives will be the same across the affected iwi, hapū and whānau. For clarity, we did not complete this review in consultation with the iwi or hapū identified in this report.

8.2 Findings

8.2.1 Terminology

Understanding more about how iwi, hapū, and whānau refer to kaimoana species, the language they are using, and their associated cultural values, uses, and practices (e.g., tohu, maramataka, manaakitanga, kaitiakitanga, rāhui) is an essential starting point when working alongside Māori communities (e.g., Wilson et al. 2007). Many non-Māori discourses have simplified the term “kaimoana” to represent coastal fisheries and/or shellfisheries; however, iwi, hapū, and whānau consider a much wider range of species when they use this language. For example, kaimoana, mahinga kai, and mahinga mātaitai are synonymous terms used by whānau, hapū, and iwi. When iwi/hapū/whānau talk about “kaimoana” this can include birds (e.g., tītī), mammals (e.g., seals), fish (e.g., blue cod), shellfish (e.g., tuangi/tuaki, pipi, mussels, pāua, oysters), crustaceans (kōura), freshwater fish (tuna), and plants/algae (e.g., karengo) (Table 7).

Table 7: Some examples of the species that may be included when iwi, hapū, and whānau refer to kaimoana. This is not an exhaustive list. (References include Wilson et al. 2007, Tipa et al. 2010a, McCarthy et al. 2014)

Birds	Mammals	Fish	Shellfish	Crustaceans	Plants
Titi	Seals	Barracouta	Kina	Crab	Parengo
		Blue cod	Kuku/kutai	Kōura	Karengo
		Conger eel	Oysters		Rimurapa
		Frostfish	Pāua		
		Groper	Pipi		
		Maomao	Pupu		
		Moki	Scallops		
		Mullet	Toheroa		
		Īnaka	Tuatua		
		Kanakana	Tuangi/tuaki		
		Kahawai			
		Kingfish			
		Ling			
		Patiki			
		Shark			
		Snapper			
		Tuna/freshwater eels			
		Warehou			
		Wheke			

8.2.2 Examples of cultural values, uses, and practices

There are common principles and values that establish and reinforce whānau, hapū, rūnanga, and iwi identity and their responsibilities and rights to manage and use kaimoana. Te Ao Māori requires an **intergenerational focus**, where resources must be protected and enhanced for those generations not yet with us and in respect of those that have passed.

Whakapapa (genealogies) anchors mātauranga Māori and provides the linked sequence and order of kinship between ngā Atua Māori (male and female deities) and the natural world. Collectively these relationships are known as whakapapa, and they represent how tangata whenua place themselves within the world, and how they understand and interact with everything around them. Oceans, estuaries, rivers, lakes, and streams are intimately bound to tangata whenua through whakapapa and are a fundamental tenant of **personal and tribal identity**. For coastal Māori, harbours, coasts and the ocean or the domain of Tangaroa (male guardian) and Hinemoana (female guardian) are considered to be the origin and source of all life, of tribal identity, whakapapa, **rangatiratanga** (authority), **mana** (prestige, authority, status), and **kaitiakitanga** (active guardianship). Islands, pinnacles, rocks, and water bodies are named after important ancestors and/or events, and the knowledge, connection, and responsibilities for those places are passed through the generations.

Māori have a special relationship with Papatūānuku (the Earth Mother) and her resources – as an integral part of the natural order and as recipients of her bounty, rather than as controllers and exploiters of their environment. Papatūānuku is to be treated with reverence, love, and responsibility rather than abuse and misuse (Marsden 2003). Therefore, for tangata whenua the environment and associated natural resources are **taonga** (significant treasure), and how they engage with it is crucial to their integrity, sense of unique culture, and ongoing ability to keep **tikanga** (correct procedure, custom, lore) and practices alive.

To sustain their mana (prestige, authority, status), kaitiaki (guardians) are bound to do everything they can to preserve and restore the **mauri** of their environment. Mauri, an internal energy or life force derived from whakapapa, is an essential essence sustaining all forms of life. Thus, it provides life, vitality, and energy to all living things and is the binding force that links the physical to the spiritual worlds. It denotes a health and spirit that permeates all living and non-living things and damage or contamination to the environment is therefore damage to or loss of mauri (Awatere & Harmsworth 2014).

The Māori world view considers the well-being of natural resources to be directly related to the **well-being of the people** (Ministry for the Environment [MfE] & Stats NZ 2019). For example, Ngāi Tahu strongly believe that the whenua and tangata are inextricably intertwined, and, when one of these becomes unbalanced, the other equally suffers. Therefore, Ngāi Tahu associate their well-being as individuals and as members of whānau, hapū, and iwi with maintaining the health of the natural environment (Sims &Thompson-Fawcett 2002, Panelli & Tipa 2007, 2009, Mahaanui Kura Taiao (2013) which is consistent with conceptualisations of well-being proposed by other indigenous communities (e.g., McGregor et al. 2003, McLennan et al. 2004).

In Te Ao Māori there are recognised transcendental values such as **manaakitanga** (care for our environment, waterways, and people), which translate directly to the way in which we manage, interact with, and perceive our marine ecosystems. For example, at a localised level there are specific marine animals, places, or practices that are integral to the contextual values of a particular whānau, hapū, or iwi (e.g., Wakefield & Walker 2005, Wilson et al. 2007). Estuaries provide kaimoana as nourishment not just for the local hapū, but also for neighbouring and inland communities (e.g., Penny et al. 2007a & b, Environs Holdings Ltd. 2011, Dalton et al. 2013). Having the ability to manaaki visitors by supplying kai sourced from one's area means that the activities of fishing and gathering other foods create and maintain community ties and reinforce identity. Conversely, the inability to manaaki guests and sustain whanaungatanga can lead to cultural loss (Tipa et al. 2010a & b).

Mātauranga Māori is specific to tangata whenua and a taonga protected by Te Tiriti o Waitangi and through legislative instruments including the RMA provisions that relate to Māori relationships with their taonga[11]. Mātauranga Māori encompasses metaphysical and esoteric forms of knowing (Marsden 2003, Mead 2003, Royal 2009a, b, & c, Roberts 2013, King 2018). Hikuroa (2017) describes mātauranga Māori as the pursuit of knowledge and comprehension of Te Taiao (the natural world) following a systematic methodology based on evidence and incorporating culture, values, and world view. Ataria et al. (2018) further explain that mātauranga Māori includes knowledge thought to be lost and now currently under recovery, with each new generation adding, subtracting, or amending the knowledge. Mātauranga Māori is developed and transmitted through practices of food management, harvesting, and preparation (Tipa et al. 2010b). Kaimoana gathering requires whānau to remember or seek the appropriate mātauranga to understand how to gather kai and resources in a manner that is respectful and consistent with their tikanga and kawa. Whānau also have to remember or learn which parts of the species are good to eat at what times of the **maramataka**, along with learning how to gather in a way that promotes **sustainability** and to how to process and prepare these foods to enhance their nutritional value. Actively practising these skills strengthens **intergenerational knowledge transfer** and, in turn, cultural, mental, and spiritual well-being.

Ki uta ki tai is an environmental philosophy that acknowledges the connectivity 'from inland/mountain to sea' and the reciprocal relationship between people and environment, which is inherent within Te Ao Māori/the Māori worldview (Te Rūnanga o Ngāi Tahu 2003, Tipa et al. 2016). Ki uta ki tai is about standing on the land and knowing the effects, both positive and negative, in every direction (Cain 2019). As Muriel Johnstone noted during a Council Hearing:

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[11] Sections 6(e), 6(f), and 6(g) require that the relationship of Māori and their culture and traditions with their ancestral lands, water, sites, wāhi tapu, and other taonga are recognised and provided for.

> "*there is no use having pristine beautiful mountains where we have made a mess of the whenua that the rivers and the waters have to run over to actually come down to the sea*."[12]

Ki uta ki tai reflects the mātauranga that all environmental elements are connected and must be managed as such. Ngāi Tahu understands ki uta ki tai as (Te Rūnanga o Ngāi Tahu 2003):

> "*a paradigm and an ethic. It's a way of understanding the natural environment, including how it functions, how people related to it and how it can be looked after appropriately…*
> *Ki uta ki tai gives reference to the Ngāi Tahu understanding of the natural world and the belief that all things are connected – a belief shared by many other iwi and indigenous people. It also highlights the central importance of mahinga kai, the traditional seasonal food gathering rituals of Ngāi Tahu and the role this played in the traditional understanding and management of natural resources.*
> *While being founded on traditional values and understanding, ki uta ki tai is also a modern management framework that involves the creation of a number of tools, such as natural resource management plans, monitoring and reporting processes and resource inventories and their associated strategies to address the continuing challenges and threats faced by all aspects of the natural environment from the mountains to the sea – ki uta, ki tai.*
> *Ki uta ki tai, as a concept, comes from the traditions, customs and values of Ngāi Tahu Whānui in relation to the natural environment, and in particular the custom of mahinga kai and transferred between generations through purākau, whakataukī, waiata, 113essoe and on-going practices is the foundation upon which this modern Ngāi Tahu natural resource management framework is built*".

8.3 Impacts of land-based stressors on kaimoana and associated cultural values, uses, and practices

8.3.1 Iwi/hapū perspectives

The direct, indirect, and cumulative impacts of land-based stressors on kaimoana and associated cultural values, uses, and practices are clearly recognised and expressed through numerous plans, reports, briefs of evidence, and other iwi/hapū-informed literature. These concerns have been voiced by iwi and hapū through various forums for a considerable period. In one example, the events leading up to, and the Motunui-Waitara claim to the Waitangi Tribunal (Waitangi Tribunal 1983), was explicitly about Te Atiawa significant concerns for their fisheries and their associated ecosystems due to the direct inputs of untreated industrial and domestic waste into the Waitara River. For example:

> *"We believe that human waste should go back to the earth. We believe that anything to do with human waste should have nothing to do with food; clothes should be washed separately from tea towels; people should not sit on food tables; seafood should not be gathered from reefs polluted by an outfall. This belief is not just related to 'scientifically detectable' pollution; even if scientists 'proved' that an outfall was not polluting, we would be unhappy gathering seafoods from a reef near such an outfall"* (James & Pawson 1995).

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[12] Quote from Muriel Johnstone cited in the Video Evidence for the proposed Southland Water and Land Plan

The direct and indirect impacts of a variety of land-based activities on mauri, water quality, kaimoana, mahinga kai, biodiversity, and manaakitanga are understood and being experienced across the motu. For example:

> *"Water quality in waterways that were prime food gathering locations for tangata whenua [in Horowhenua] have been highly compromised. There has been associated drops in shellfish health and abundance. This is deeply concerning to local Iwi and Hapū"* (Hardy et al. 2011).
>
> *"Upstream discharges to water whether from point or non-point sources lead to increased nutrient and contaminant loads which in turn degrade downstream water quality. These discharges impact the life supporting capacity of estuarine and seawater area; they impact kaimoana, kaimātaitai and mahinga kai; and impact cultural use of estuaries and the ocean"* (Ngāi Tahu ki Murihiku 2008).
>
> *"The mauri of water [including freshwater, estuarine and coastal environs] and soil and their associated ecosystem within rohe of Ngātiwai is being destroyed or lost through ignorance, oversight, misuse, exploitation, contamination and abuse"* (IEMP Ngātiwai).
>
> *"With the loss of ability to manaaki (care for) guests at the marae with local delicacies. For example, with the dramatic decline of species such as toheroa (Paphies ventricosa) and tohemanga (Longimactra elongate), there are concerns that these culturally important resources will not ever recover sufficiently for sustainable harvest"* (Smith et al. 2011).
>
> *"The ability to put kaimoana on the table for manuhiri and whānau at tangi, hui and other events on Ngātiwai marae, and to feed Ngātiwai whānau and hapū on a regular, sustained basis, is being increasingly compromised by damage to the mauri of the water"* (IEMP Ngātiwai).

Whānau experiences of how land-to-sea stressors have impacted kaimoana, and the associated impacts on their cultural identify and mana, have long been voiced across the motu (Table 8). The issue of sediments and sedimentation is specifically mentioned in numerous sources. For example:

> *"Tangata whenua have been concerned for some time about the degradation of coastal resources, the loss of kaimoana (seafood) or the increasing toxicity of remaining marine species, sedimentation, pollution, eutrophication of waterways, and the associated negative impact that such issues have on cultural identity and sense of pride (mana)"* (Hardy et al. 2011).
>
> *"Within Tauranga, sedimentation is a frequently cited concern from the local community, including Iwi and Hapū"* (van den Belt et al. 2012, Taiapa et al. 2014).
>
> *"Although estuarine shellfish have long been important mahinga kai for whānau within Murihiku (Ngāi Tahu Claims Settlement Act 1998b), shellfish sites such as pipi and tuaki in Koreti Waimātaitai, are now covered by deep soft muds"* (Robertson & Stevens 2010).

It is recognised that sediment is associated with other contaminants such as phosphorus and heavy metals which may impact kaimoana, mahinga kai, and human health. Collaborative research with Te Arawa, Arowhenua, and Te Waihora kai gatherers confirmed that sediment was identified as an issue by whānau (Tipa et al. 2010a & b, Stewart et al. 2014) and that there were potential health risks for the consumption of some freshwater and coastal wild-caught species even at low consumption rates (Stewart et al. 2011, Phillips et al. 2014, Stewart et al. 2014). Contaminant concentrations in shellfisheries and fisheries are a concern when they affect food safety and socio-cultural wellbeing (e.g., Fisher & Vallance 2010, Phillips et al. 2011, King & Lake 2013, Kainamu-Murchie 2017, Kitson 2017). For example:

> *"There are specific areas in the harbour where nutrient and other pollutants discharges are of concern at times, particularly when shellfish contamination occurs, making them unsafe for human consumption"* (Hardy et al. 2011).

8.3.2 Place-based case study – Waikato IEMPs

Huirama (2021) completed a review of IEMPs from the Waikato region to better understand hapū and iwi knowledge of sediment-related issues and impacts, and the responses/actions suggested by hapū/iwi to address these issues (Figure 51). Land use and associated land management practices, soil degradation, and resulting impacts on mahinga kai and water quality/clarity are key issues included in these IEMPs. These plans express that activities on the land are interconnected and impact freshwater, estuarine, and coastal ecosystems.

Hapū and iwi are concerned about the degradation of the mauri of the land due to erosion and the impacts of sediment on the mauri of freshwater and marine environments. Specific land uses of concern were agriculture, forestry, and urban development. Agricultural land use issues (e.g., sheep and beef, dairy) included vegetation clearance for farming causing increased sediment run-off, the transportation of phosphorus (P) with eroded soil, access of stock to waterways, and decreasing water quality (e.g., disturbing sediments and releasing P). Forestry issues were associated with poor management practices (e.g., the increase of erosion and sediment run-off, during and after harvesting). Urban development issues included the lack of appropriate controls during earthworks (e.g., establishment of transport networks).

Other notable sediment sources mentioned were stream bank erosion in relation to land-use practices (e.g., farming, forestry, and urban development/earthworks) which are the perceived drivers of erosion, alongside associated land management practices (e.g., vegetation clearance, stock damage). The IEMPs mentioned various erosion types of concern including stream bank erosion, hillslope erosion, mass movements, and coastal erosion. Stream bank erosion issues included bank erosion threatening the location of a marae, whānau observations of rivers widening over time, and the exposure of kōiwi in riverbanks, on the coast, and in paddocks. Vegetation clearance was mentioned as the key driver of hillslope erosion and mass movement. The additional impacts of vegetation clearance not only include the degradation of ecosystems (i.e., fragmentation of the ngahere) but also cultural/spiritual impacts on Papatūānuku and whānau/hapū/iwi.

Water quality and mahinga kai were key values identified by hapū and iwi as being impacted by sediment, in both freshwater and receiving environments (i.e., estuarine/coastal). Sediment impacts localised sources of kai, including kaimoana, and the ability to see kai for harvesting. For example, suspended sediment impacts water clarity which in turn impacts the ability of taonga species to forage, recreational values (swimming, bathing), and cultural harvesting (e.g., ability to access the site). It also increases the time needed to clean harvest equipment (e.g., nets).

Table 8: Examples of land-based stressors and the direct and indirect impacts on estuarine and coastal kaimoana as expressed by iwi and hapū

Stressors identified	Description	Examples of iwi/hapū that have described these issues and their impacts
Land use, land management, coastal erosion	Perceived drivers of coastal erosion include land use changes (e.g., vegetation clearance near the coast) subsequently increasing sedimentation of estuaries. This is linked to cultural/spiritual impacts on mahinga kai, wāhi tapu, and sites of significance. Impacts of land management activities (e.g., drained wetlands, wastewater/industrial inputs) and unsustainable fishing practices are specifically linked to declines in kaimoana.	Ngāti Maniapoto, Hauraki Whānui, Ngāti Hikairo, Ngāti Tahinga, Waikato-Tainui, Ngāi Tahu ki Otakou, Ngāi Tahu ki Murihiku.
Sedimentation	Sedimentation is recognised to reduce water clarity, clog the gills of filter feeders (for example cockles, pipi, scallops), reduce the foraging abilities of finfish (e.g., juvenile snapper), impacting reefs, seaweeds, kina, nursery grounds and decreasing food available to benthic species. Specific locations are mentioned that are changing from sandy to muddy, impacting kaimoana and the ability of whānau to interact with these locations.	Hauraki Whānui, Waikato-Tainui, Ngāi Tahu ki Waitaha, Mootakotako Marae Hapū, Ngāti Maniapoto, Tahinga, Waikato-Tainui, Ngāi Tahu ki Otakou
Nutrients	Water quality issues such as excessive nutrients, algal blooms, and eutrophication are issues that impact fisheries/kaimoana, their habitats and the ability of whānau to interact safely with these locations. Restrictions on shellfish harvest (rāhui and closures) due to eutrophication and toxic algal species occur around Aotearoa New Zealand.	Hauraki Whānui, Ngāti Porou ki Hauraki, Tahinga, Ngāti Awa – Tauranga, Ngāi Tahu ki Murihiku
Point source discharges	Point source discharges (e.g., sewage, wastewater, stormwater) are issues that impact mauri, mahinga kai, and biodiversity, as well as the ability of whānau to safety interact with these locations. Sewage from non-functioning pumping stations and old pipelines are identified as impacting kaimoana. Iwi and hapū are averse to taking food from polluted waters, especially water bodies receiving treated and untreated sewage.	Ngāi Tahu ki Waitaha; Ngāi Tahu ki Murihiku (IEMP)
Heavy metals and other contaminants	Sediment entering the environment, for example through stormwater discharges, is known to carry contaminants (e.g., heavy metals) that impacts fishery (and potentially human) health and customary fishery practices.	Hauraki Whānui, Mootakotako Marae Hapū, Ngāti Maniapoto, Tahinga, Waikato-Tainui, Ngātiwai, Ngāti Awa – Tauranga, Ngāi Tahu ki Waitaha, Ngāi Tahu ki Murihiku, Ngāi Tahu ki Otakou.

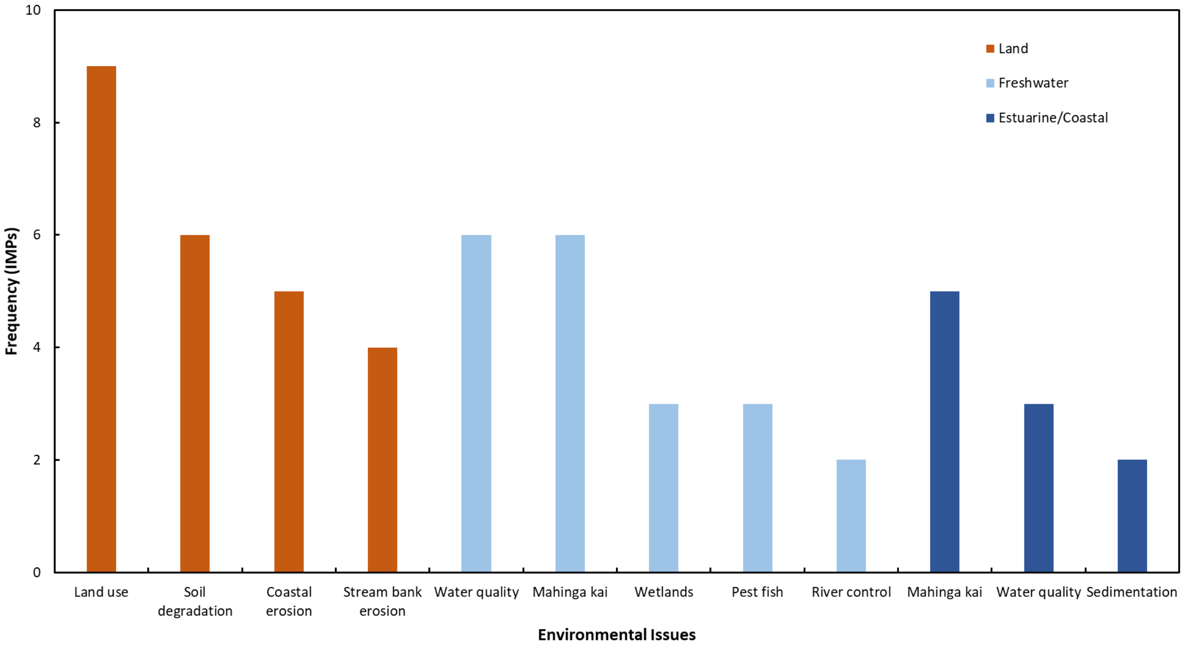


Figure 51: Erosion and sediment issues identified by hapū and iwi in the Waikato Region for land, freshwater, and estuarine/coastal environs. (Source: Huirama 2021.)

8.3.3 Place-based case study – Hauraki

Whaia te Mahere Taiao a Hauraki is a strategy for collective action by Hauraki whānui to sustain the mauri of the natural environment and cultural heritage of the Hauraki rohe over the next 50 years (Hauraki Māori Trust Board 2004). This plan seeks to return to a situation where fisheries and bird life are more abundant, and the waters are clearer and cleaner. The goals and environmental dependencies of various cultural values, uses, and practices expressed in the vision of Whaia te Mahere Taiao a Hauraki (bolded) are that "*...our waterways in rural and urban areas have been* ***restored to their indigenous state****, and are now* ***home to increasing populations of fish****.* ***Water plants have increased*** *providing a home for kaeo and protecting young of tuna and inanga. There are* ***no longer any contaminants polluting our waterways and harbours****, and many* ***more wetlands exist*** *because of the past and present activities of kaitiaki...These* ***places of the land and sea have once again become abundant food baskets****.* ***Kaitiaki are protecting the harvesting of mahinga kai, mahinga mātaitai*** *and taonga raranga for tribal communities in accordance with tikanga...*".

Of relevance to this review, this plan reflects that the various atua domains are interconnected as are the groundwaters, fresh waters, and coastal waters. Pollution entering coastal waters from marine, freshwater and groundwater sources have impacted Pare Hauraki Iwi cultural values, uses, and practices. The impacts of land-based activities, including sedimentation, feature strongly in the drivers/pressures outlined in this plan including:

- Papatūānuku: land use change – resulting in loss of wetlands, peat lands, and coastal dune lands and increased sedimentation; land use intensification and management, including landfills, farming, and industry – resulting in contaminants in waterways (e.g., nutrients, toxins, sewerage), erosion, and sedimentation.
- Tangaroa Rerenga Wai Tai: marine currents – resulting in nutrients and other contaminants; maritime transport – resulting in oil and chemical spills, ballast water, sewerage, plastic debris, other waste; land-use management – resulting in sedimentation on reefs and seaweed affecting juvenile settlement for kina and pāua, and the extent and abundance of kaimoana; commercial and recreational harvesting – reducing the extent and abundance of kaimoana.

- Tangaroa Rerenga Wai Māori: river straightening, drainage, and reclamation – irreversibly modifying estuaries and coastal lagoons, their salt marshes and mangrove forests, turning harbours and coastal bays from sandy to muddy areas.
- Ranginui: maritime transport – resulting in air pollution (fuel emissions). Understanding the implications of climate change is also signalled in this plan.

The role of coastal sediments as a sink for contaminants is recognised in this plan as are the secondary impacts of activities such as dredging which may resuspend legacy contaminants into the habitats, water, and food chains supporting various kaimoana/taonga species. Various taonga species with life-cycle dependencies associated with the habitat and water quality of both fresh waters and coastal waters are specifically mentioned in this plan, including tuna (juvenile and migrant freshwater eels), īnanga/kōkopu/whitebait, and kāeo (freshwater mussels, whose life cycle is in turn dependent on diadromous fish species). The on-going decline of a wide variety of kaimoana species is specifically highlighted in the plan including snapper, kina, pāua, kōura, kūtai, pipi, tuangi, and tipa. Each of these species will have specific habitat and water quality requirements during each stage of their life cycle. Whales, dolphins, and seals are also covered in this plan.

The time frames covered by this plan are intergenerational and the knowledge and tools kaitiaki are seeking to achieve their vision may come from a variety of knowledge systems, i.e., "*Hauraki people are making final decisions about change in the environment through the eyes of the next generation. This is because our mokopuna have been taught what it is to be kaitiaki and the tikanga about how you manaaki the environment from their whaea, matua and whānau. They also freely take the tools from the western science to enhance their kaitiaki activities*".

8.3.4 Place-based case study – Waitaha (Canterbury region)

Kainamu-Murchie et al. (2018) interviewed local Ngāi Tahu fishers, recreational fishers and 'beach-goers' to better understand their experiences in the condition of their estuaries and shellfisheries overtime. This study showed that perceptions around sediment condition, water condition, and/or waterway input (sewage/organic waste), as well other contaminants, were used by fishers as indicators that informed their place-based interactions with four estuaries – Rakahuri-Saltwater Creek Estuary, the Avon-Heathcote Ihutai Estuary, Rāpaki Bay, and Koukourarata.

Experienced fishers, including Ngāi Tahu, changed or ceased their mahinga kai practices when environmental conditions were degraded, as indicated by declining abundances in mahinga kai and the presence of anthropogenic and/or natural hazards (e.g., earthquake impacts) (Kainamu-Murchie 2017). As a result, in some locations, favoured species, such as tuaki/cockles, were no longer harvested due to perceived decreased abundances, gathering restrictions (rāhui), and/or poor environmental conditions (e.g., food-safety risks) (Kainamu-Murchie et al. 2018). This study highlighted that land-based stressors were impacting site condition and fisher activities. For example:

> *"Food gathering has diminished, and it is polluted. We are not allowed to gather because it is not safe"*
>
> *"It is silty; Shellfish beds have become smaller and smaller"*

The negative impact of sediment (especially silt) on shellfish/kaimoana is a concern for Koukourarata (Kainamu-Murchie et al. 2018, Mahaanui Kurataiao Ltd 2013) and Ngāti Wheke kaitiaki (Mudunaivalu 2013, Mahaanui Kurataiao Ltd 2013). Due to their concerns about declining shellfish populations, mātaitai reserves were established at Koukourarata and Rāpaki Bay to recognise the cultural importance of these sites, to exercise their kaitiakitanga, and to provide food gathering areas (Mudunaivalu 2013). Similarly, Te Whakaraupō/Lyttelton Harbour was once used as mahinga kai, however, by the early 1990s very little food was gathered from the beach or harbour due to effluent discharges and sedimentation (Tau et al. 1992).

8.3.5 Place-based case study – Aparima

Aparima (Jacobs River) Estuary is regarded by Ngāi Tahu ki Murihiku whānau as an important mahinga kai with shellfish, tuna, and īnaka gathered from the river and estuary (Ngāi Tahu Claims Settlement Act 1998b, Te Ao Marama Inc 2020). Of significance, within the estuary there was an area known as Marakura that comprised several hectares of marine plants (e.g., seaweed and seagrass) and shellfish resources that were exposed during low tide (Cain 2016).

The Aparima River is a Statutory Acknowledgement Area under the Ngāi Tahu Settlement Act 1998 (NTCSA). This statutory instrument records the significance of permanent settlement at the river mouth with urupā (burial sites) located in this area. Despite this Aparima Estuary is suffering degradation, due to a combination of rural and urban contaminant sources, including faecal contamination, nutrients, sediments, and heavy metals. Given its cultural significance, efforts to prevent further degradation and restore the estuary are a priority for mana whenua (Te Ao Marama Inc 2020).

There is a need for waimātaitai-specific research and the development of culturally appropriate methodologies to monitor the state and pressures on estuarine cultural values and associated uses, e.g., mahinga kai. A recent project between Te Rūnaka o Ōraka-Aparima and NIWA explored the dependencies between Aparima waimātaitai condition, and the cultural values and uses of Ōraka-Aparima whānau. It drew on mātauranga Māori and science to identify culturally appropriate indicators that informed the design of a monitoring approach to assess the state of two priority cultural values – tuaki and pātiki (Kainamu et al. in prep). As part of this project, the latest knowledge was collated to provide an overview of the pressures on tuaki and pātiki populations within the Murihiku rohe (Figure 52) (Kainamu-Murchie et al. 2019).

Changes to Aparima Estuary (associated with changes to wetlands, waterway, and sediment condition) were identified by Ngāi Tahu ki Murihiku whānau to have impacted ecology of Aparima waimātaitai and their associated practices. Wetlands that once supported raupō, harakeke, and fisheries spawning grounds, as well as likely providing other services such as nutrient and sediment retention or filtering, have declined. Increases in sediment and resulting changes to the substrate were associated with a wide range of Ngāi Tahu ki Murihiku whānau concerns. Recent studies have suggested extensive and increasing eutrophication (very low sediment oxygenation and sulphide-rich sediments) and sedimentation problems (Robertson & Stevens 2012, 2013). Changes in sediment quality (including colour and smell) is a concern for whānau which is perceived to be associated with contaminants and is influencing their interactions with the estuary (Kainamu et al. in prep).

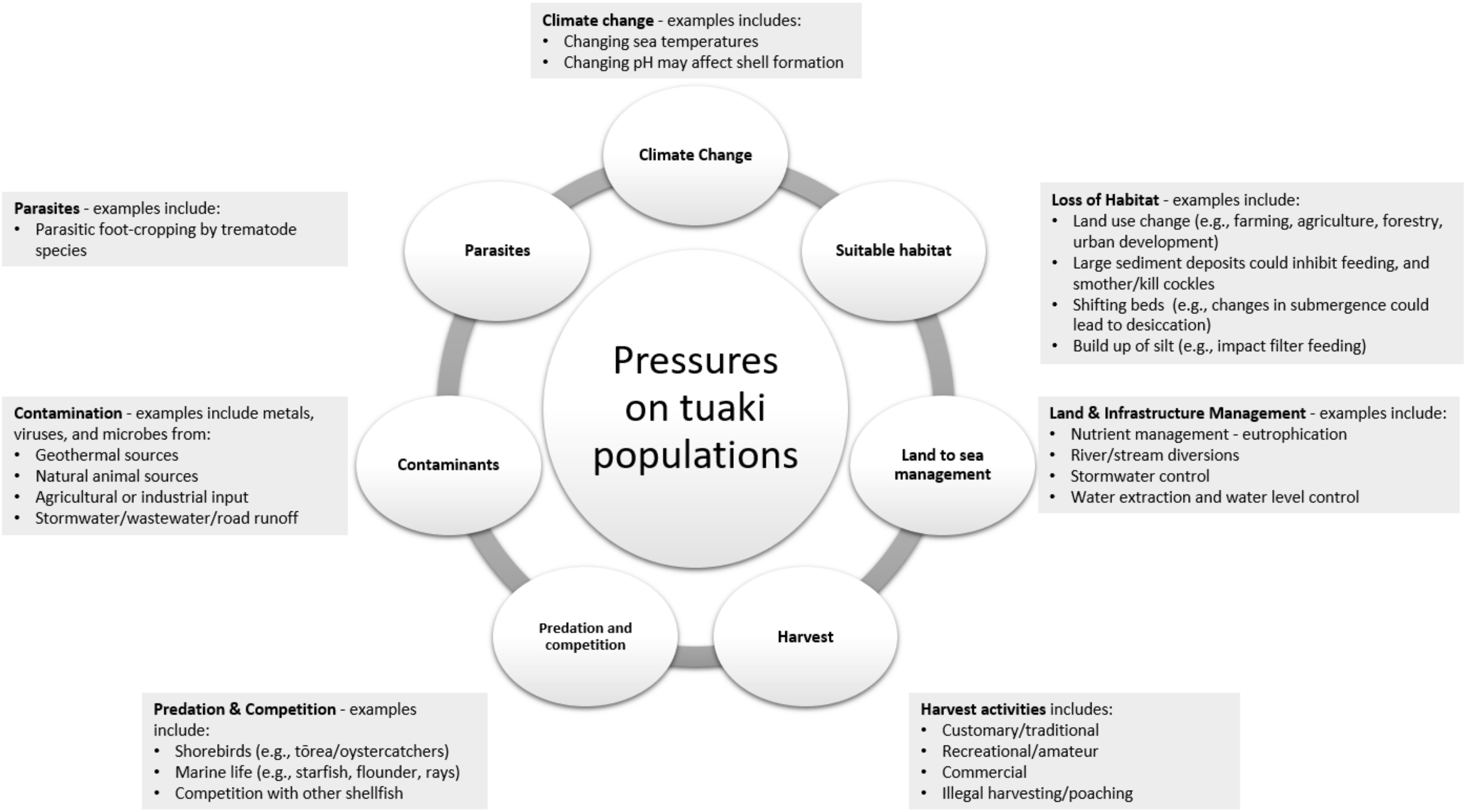
Although Murihiku whānau have access to valuable intergenerational mātauranga and experiences, this study identified that monitoring by Environment Southland did not include any sites or times of importance to Murihiku whānau – and although it is a key concern for whanau, mahinga kai food safety is not assessed in this estuary. Murihiku whānau want to better understand how sedimentation impacts tuaki and pātiki condition and population structure over time. They want to know how changes in sediment influence water quality and their ability to interact with these sites safely. They also want to know if their taonga species are safe for consumption. To this end, Ngāi Tahu ki Murihiku have recently started working on these issues with Environment Southland in the Aparima estuary. They have installed the first sediment plates in areas of relevance to them for long-term monitoring by the council and rūnaka (Kainamu et al. in prep).

8.3.6 Discussion

In this review of selected literature, common and recurring land-based stressors on kaimoana species, and their associated environments and cultural practices – as expressed by iwi and hapū – included land use, land management, sedimentation, nutrients, discharges, and contaminants (including sediments/substrates acting as a source and sink) (Table 8). The sedimentation that has been observed by iwi and hapū is changing the composition of estuarine habitats and making them muddier. Sediment impacts water quality/clarity and degrades the mauri of the marine environment, and it impacts cultural (e.g., kai, harvesting, medicine) and amenity values (e.g., sediment plumes making the harbours look turbid). Land-based stressors, including sediment, impacts mahinga kai health (e.g., heavy metal contamination) and ecosystem habitats (e.g., nursery grounds for marine species like fin fish, reefs, seaweeds, kina, shellfish beds).

Each hapū and iwi have their own mātauranga which are specific and relative to their environmental contexts, experiences, observations, and understandings of their interactions and patterns of use (e.g., Wakefield & Walker 2005, Wilson et al. 2007, Paul-Burke & Rameka 2015, Crow et al. 2018; Hepi et al. 2018, Jackson et al. 2018, Kitson et al. 2018, Maxwell et al. 2018, Ratana et al. 2019). These accumulated intergenerational understandings and practices are grounded in the existence of Māori/tangata whenua, which are intimately bound to residing in one place for many generations (e.g., Cheung 2008). Land-based stressors have had direct, indirect, and cumulative impacts on the abundance, distribution, and condition of kaimoana and also the wellbeing of whānau, hapū, and iwi for whom these species are important for their cultural identity and survival (Figure 53). In some cases, these impacts have been severe (e.g., McCarthy et al. 2014).

Iwi and hapū resource management systems are important for the sustainability of the environment and its resources and must also be understood by agencies (e.g., Garven et al. 1997, Wilson et al 2007). Kaimoana gathering is the glue that binds whānau, hapū, and community together, providing a sense of identity that also serves as the vehicle for the transmission of values and knowledge (Tipa et al. 2010a). Environmental degradation and consequent impacts on mahinga kai species and cultural practices is a significant issue for iwi and hapū across Aotearoa New Zealand. They are also experiencing impacts through the loss of mātauranga and their cultural identify – therefore, at stake with the loss of kai gathering is not only cultural survival, but potentially the physical and mental well-being of whānau (Tipa et al. 2010a).



Climate change - examples includes:
- Changing sea temperatures
- Changing pH may affect shell formation

Parasites - examples include:
- Parasitic foot-cropping by trematode species

Loss of Habitat - examples include:
- Land use change (e.g., farming, agriculture, forestry, urban development)
- Large sediment deposits could inhibit feeding, and smother/kill cockles
- Shifting beds (e.g., changes in submergence could lead to desiccation)
- Build up of silt (e.g., impact filter feeding)

Contamination - examples include metals, viruses, and microbes from:
- Geothermal sources
- Natural animal sources
- Agricultural or industrial input
- Stormwater/wastewater/road runoff

Land & Infrastructure Management - examples include:
- Nutrient management - eutrophication
- River/stream diversions
- Stormwater control
- Water extraction and water level control

Predation & Competition - examples include:
- Shorebirds (e.g., tōrea/oystercatchers)
- Marine life (e.g., starfish, flounder, rays)
- Competition with other shellfish

Harvest activities includes:
- Customary/traditional
- Recreational/amateur
- Commercial
- Illegal harvesting/poaching

Figure 52: Examples of some of the pressures on tuaki (*A. stutchburyi*) populations. The pressures above are inclusive of those known to impact tuaki populations across Aotearoa New Zealand and are not restricted to Murihiku. (Source: Kainamu-Murchie et al. 2019.)

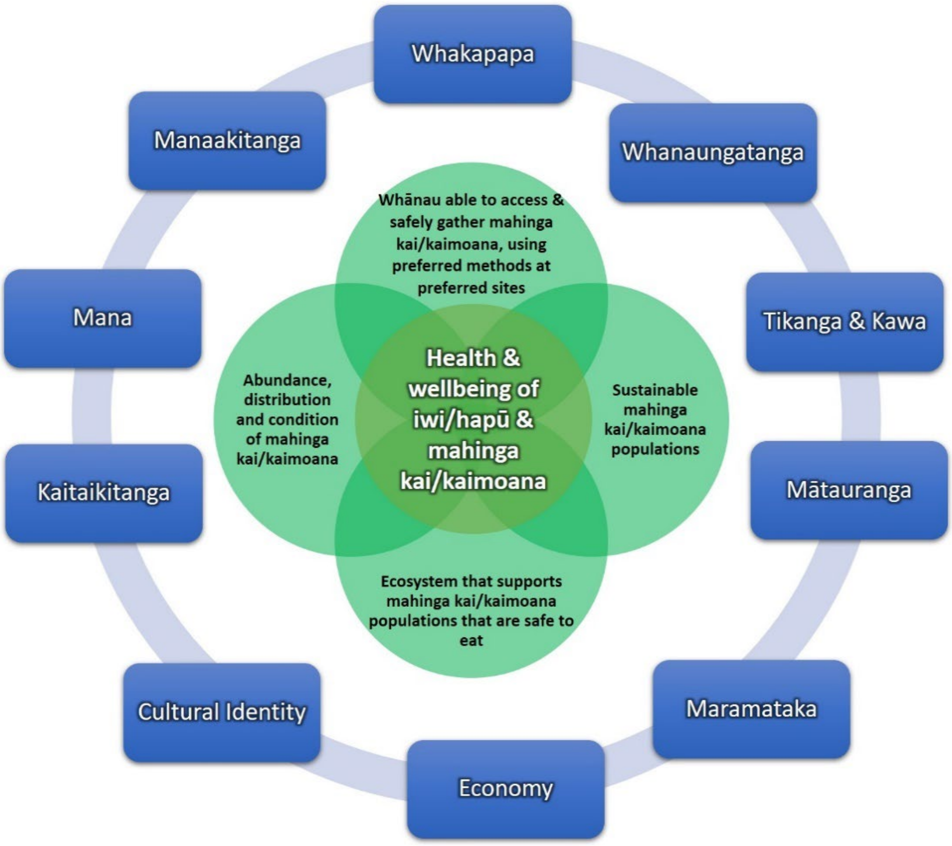


Figure 53: Land-based stressors, including sedimentation, impact kaimoana and a variety of associated cultural principles, values, uses, and practices. This review has demonstrated that these issues have been experienced by iwi and hapū throughout Aotearoa New Zealand for a long period of time. In some cases, these impacts have been severe.

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

With present knowledge of land-based impacts on fisheries in New Zealand still 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 arose 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, pāua, 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 impacts 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, Whangārei, Waitematā, and Tauranga harbours) and reductions in the availability of harvested species such as cockles and pipi from areas 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 indeed, any exists at all.

10. GAP ANALYSIS AND SUGGESTIONS

Our current understanding of land-based impacts on coastal fisheries and supporting biodiversity in New Zealand remains very limited. New research complete since the previous 2009 review has added to our fundamental knowledge of how land-based stressors are acting and emphasises that impacts are ecologically, economically, and socially important. Society's increasing realisation of the importance of mātauranga Māori is also coming to the fore, emphasising the disproportionate consequences of land-based impacts on some communities.

While some progress has been made, the broad generic research questions suggested by Morrison et al. (2009) remain centrally relevant in 2021. These need to be addressed in combination with an examination of the relative role of marine-based stressors, such as fishing. They include:

- FISH–HABITAT ASSOCIATIONS: Fundamental and systematic inventories 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 fishery 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: e.g., the 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 each 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 remain unexplored in New Zealand and deserve attention. Ideally, species and areas should be selected within a framework that 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-derived 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 to 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.

To these we would add:

- THE INFLUENCE OF SUBMARINE GROUNDWATER DISCHARGE (SGD): Evidence is increasing on the important role these inputs may play in ecological and fisheries productivity in some coastal systems. With large changes in weather systems, including rainfall with climate change, the ever-increasing abstraction of fresh water from aquifers and surface water sources by humans, and rising sea levels, the value provided by SGD to New Zealand's fisheries (currently unknown) is highly like to change in the future.

11. ACKNOWLEDGEMENTS

We thank Fisheries New Zealand for commissioning this work under project ZBD2020-11, and Jonathan Moores for providing much appreciated editorial comment.

12. GLOSSARY

Atua	Ancestry with continuing influence, god, deity – although often translated as 'god' and now also used for the Christian God, this is a misconception of the real meaning. They are regarded as ancestors with influence over particular domains and related to man.
Hapū	Kinship group, clan, tribe, sub-tribe, extended family – often refers to a sub-tribal/extended family kinship group, that consists of extend family who descend from a common ancestor.
Iwi	Extended kinship group, tribe, nation, people, nationality – often refers to a large group of people descended from a common ancestor and associated with a district territory.
Kaitiaki/tangata tiaki	The contemporary definition is utilised in this research in regard to fisheries: the custodian, guardian, keeper, steward of customary fisheries designated by tangata whenua.
Kaitiakitanga	The intergenerational exercise of customary custodianship, in a manner that incorporates spiritual matters, by those who hold mana whenua/moana status for a particular area or resource.
Kaimoana	Kaimoana may simply represent coastal fisheries and/or shellfisheries; however, iwi, hapū, and whānau consider a much wider range of species when they use this language. For example, kaimoana, mahinga kai, and mahinga mātaitai are synonymous terms used by iwi, hapū, and whānau.
Ki uta ki tai	It is an environmental philosophy that acknowledges the connectivity 'from inland/mountain to sea' and the reciprocal relationship between people and environment.
Mahinga kai	Places at which food (and other commodities) were extracted or produced, and it also signified food items obtained at those places, the methods by which food was secured, cooked, or prepared for eating, or preserved for later use or for gift exchange.
Manaaki/manaakitanga	The practice of showing respect, generosity, and care for others, caring for people, places, and other living and non-living things.
Mana	Prestige, authority, status.
Mana whenua, mana moana	Refers to the local tribe/sub-tribal group who hold mana and have 'demonstrated authority' over land or territory in a particular area, authority which is derived through whakapapa links to that area.
Māori	Indigenous people of Aotearoa, literally. Original, normal, ordinary.
Maramataka	Māori lunar calendar – a planting and fishing monthly almanac.
Mātauranga Māori	Is a holistic perspective encompassing all aspects of Māori knowledge and seeks to understand the relationships between all component parts and their interconnections to gain an understanding of the whole system. It is based on its own principles, frameworks, classification systems, explanations, and terminology. It captures both traditional knowledge as well as new knowledge being created every day in Māori communities. Mātauranga Māori is a dynamic and evolving knowledge system, has both qualitative and quantitative aspects, and includes the processes for acquiring, managing, applying, and transferring that body of knowledge.
Mātauranga a-iwi/hapū	Knowledge systems held specifically by iwi and hapū.
Mauri	Life principle, life force, vital essence, special nature, a material symbol of a life principle, source of emotions – the essential quality and vitality of a being or entity. Also used for a physical object, individual, ecosystem, or social group in which this essence is located.
Rangatiratanga	Self-determination, sovereignty, autonomy, self-government, domination, rule, control, power. For instance, the capacity of hapū and iwi to exercise authority over their own affairs. The Crown has the duty to recognise hapū and iwi rights to exercise this authority and self-determination.
Rūnaka/Rūnanga/ Papatipu Rūnanga	This is the customary/tribal/sub-tribal assembly or council. In a Ngāi Tahu context, there are 18 Papatipu Rūnanga that exist to uphold the mana of their region. Each rūnanga has its own governance structure and appoints a tribal member to represent its interest at Te Rūnanga o Ngāi Tahu, the governing council overseeing the tribe's activities.

Rāhui	A closure to harvesting and/or activities within a particular site, due to, but not limited to, health and environmental disturbances (i.e., the earthquake and its associated sewer impacts), any incidents from being in the sea, or marine habitat restoration purposes.
Tangata tiaki/kaitiaki	The contemporary definition is utilised in this research in relation to fisheries: the custodian, guardian, keeper, steward of customary fisheries designated by tangata whenua.
Tangata whenua	Indigenous people of Aotearoa New Zealand, literally 'people of the land'.
Taonga	Treasures of cultural and historical significance to Māori, e.g., can include species of indigenous flora and fauna.
Takiwā	District, area, territory, vicinity, region.
Tikanga	Māori customary law, values, and practices. Also encompasses the correct procedure, custom, lore, method, and practice. The customary system of Māori values and practices or set of protocols that have developed over time and are deeply embedded in the social context.
Tohu	Sign, mark, symbol, cue, landmark, distinguishing feature.
Whakapapa	Genealogical or ancestral relationships/ties to person, place, or natural resource.
Whānau	An extended family, family group, or a familiar term of address to a number of people

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14. APPENDIX A: KEY NATIONAL DATASETS

Spatial data

The datasets below are those that are available nationally that can be used as input to catchment-scale water quality models. MW / LCR is Manaaki Whenua-Landcare Research; GNS is GNS Science; LINZ is Toitū Te Whenua Land Information New Zealand; MfE is Ministry for the Environment; NIWA is National Institute of Water and Atmospheric Research.

Name	**Alias**	**Datatype**	**Developer / owner**	**Description**	**Latest release**	**URL**	**Licence**
Land Cover Data Base	LCDB	Polygon shapefile	MW / LCR	Land covers derived from satellite imagery. Latest version is LCDB5 and has the reference year 2018. Divides NZ mainland into 33 land cover classes.	2020 (LCDB5)	https://lris.scinfo.org.nz/layer/104400-lcdb-v50-land-cover-database-version-50-mainland-new-zealand/	Creative Commons Attribution 4.0 International
Agribase		Polygon shapefile	Assure Quality	Farm locations and types including livestock, cropping, horticulture, viticulture and forestry. Contains information on stocking rates.	Continuously updated	https://www.asurequality.com/services/agribase/	Proprietary Cost varies by dataset options
S-map		Polygon shapefile	MW / LCR	Soil types and associated physical properties including depth, drainage, stoniness and clay content. Update to FSL.	2021	https://smap.landcareresearch.co.nz/ Data viewer and searchable database (registration required) Data download: https://lris.scinfo.org.nz/layer/48440-s-map-a-new-soil-spatial-information-system-for-new-zealand-current/ Mirror: https://koordinates.com/from/lris.scinfo.org.nz/layer/48440/ Data download	Creative Commons Attribution-NonCommercial-No Derivative Works 3.0 New Zealand Licence

Name	Alias	Datatype	Developer / owner	Description	Latest release	URL	Licence
Fundamental soil Layer	FSL	Polygon shapefile	MW / LCR	Soil types and associated physical properties including depth, drainage, stoniness and clay content. Superseded by S-map	2010	https://lris.scinfo.org.nz/layer/48079-fsl-new-zealand-soil-classification/ Mirror: https://koordinates.com/from/lris.scinfo.org.nz/layer/48079/	Landcare Data User Licence
1:250,000 Geological Map of New Zealand	Q-MAP	Polygon Shapefile or raster grid Jpeg images	GNS	Geological data including geomorphology, stratigraphy, tectonic history, geological resources, geological hazards and engineering geology	Varies by region	Data viewer: https://data.gns.cri.nz/geology/	Map images (jpegs) free to download under Creative Commons Attribution 4.0
Land Use Carbon Analysis System	LUCAS	Polygon shapefile	MfE	Regional shapefiles that map four key land use classes i.e. Natural forest, Pre-1990 Planted Forest, Post1989 Forest, and Grassland with woody biomass.	2016	https://data.mfe.govt.nz/layer/52375-lucas-nz-land-use-map-1990-2008-2012-2016-v008/	Creative Commons Attribution 4.0 International
River Environments Classification	REC	Network geodatabase	NIWA	Stream network represented by stream lines (or segments) and subcatchments. Data includes connectivity, and hydrological and catchment physical properties	2019 (REC v2.5)	https://niwa.co.nz/freshwater-and-estuaries/management-tools/river-environment-classification	
NZ 8m Digital Elevation Model (2012)		Raster	LINZ	This 8m Digital Elevation Model (DEM) was originally created by Geographx (geographx.co.nz) and was primarily derived from January 2012 LINZ Topo50 20m contours (data.linz.govt.nz/layer/768). For a full description of the how the DEM was generated refer to this layer's metadata.	2016	https://data.linz.govt.nz/layer/51768-nz-8m-digital-elevation-model-2012/ Mirror: https://koordinates.com/from/data.linz.govt.nz/layer/51768/	Creative Commons Attribution 4.0 International

Name	Alias	Datatype	Developer / owner	Description	Latest release	URL	Licence
Land Resources Information	LRI	Polygon shapefile	MW / LCR	Set of national data layers including: Land Use Capability, slope, lithology and soil (see FSL) Details given by Newsome et al. (2008)	2010	https://lris.scinfo.org.nz/layer/48076-nzlri-land-use-capability/	Landcare Data Use Licence
Virtual Climate Station Network	VCSN	Raster	NIWA	Interpolated climate maps for common climate metrics. Available as seasonal and annual summaries (latest 1991-2020) as well as daily and monthly datasets	Continuously updated	https://niwa.co.nz/climate/our-services/virtual-climate-stations Map images available from NIWA Datasets available on request	Creative Commons Attribution 4.0 International
LiDAR Digital Elevation Models		Raster	LINZ	LiDAR DEMs with 1m resolution	Varies by region	Available by region from LINZ or Koordinates Search for LiDAR DEM and region https://data.linz.govt.nz https://koordinates.com	Creative Commons Attribution 4.0 International

Non-spatial data

The following datasets include time series data and discrete point data that are publicly available and can be used as model inputs or for model calibration.

Name	Alias	Datatype	Developer / owner	Description	Latest release	URL
Virtual Climate Station Network	VCSN	Time series	NIWA	Interpolated climate data	Continuously updated	https://niwa.co.nz/climate/our-services/virtual-climate-stations Datasets available on request
Climate station data		Timeseries	NIWA	Measured climate data	Continuously updated	NIWA CLIFLO climate data portal. Registration required
National Rivers Water Quality Network	NRWQN	Discrete measurements	NIWA	Multi-variable water quality data for 77 sites	Continuously updated	https://niwa.co.nz/freshwater/water-quality-monitoring-and-advice/national-river-water-quality-network-nrwqn Online visualisation. Datasets available on request
Urban Runoff Quality Information System (URQIS)	URQIS	Online data summaries	NIWA Auckland Council	Query tool that allows uses to query an underlying urban water quality database	2012	https://niwa.co.nz/information-services/urban-runoff-quality-information-system-urqis Data summaries online. No access to underlying database
Survey of rural Decision makers	SRDM	Online	MW / LCR	Biennial survey of farmers, foresters, growers and lifestyle block owners on the their land management, economics and wellbeing etc.	2021	Online summaries and information sheets available from: https://www.landcareresearch.co.nz/discover-our-research/environment/sustainable-society-and-policy/survey-of-rural-decision-makers/srdm-2021/ Underlying data available on request.
State of Environment water quality reporting	SOE	Discrete measurements	Regional Councils, Unitary Authorities	Monthly multivariable water quality measurements made at over 1000 monitoring sites.	Continuously updated	Data availability and downloads vary by region, generally by request. Some regions have interactive data visualisation tools. Summaries of data available by monitoring site at Land and Water Aotearoa https://www.lawa.org.nz/

Model outputs

The following datasets are the outputs of water quality models that have been run at the national scale. This list is restricted to those models where the outputs are available for download. We are aware of other model applications at national and regional scale where model outputs have not been made available for download. These applications include national models for policy development and regional modelling of nutrients and sediments to support catchment and mitigation planning. Examples include both steady state models such as CLUES (Elliott et al. 2016, Semadeni-Davies et al. 2019, Semadeni-Davies et al. 2020) and SedNetNZ (Mueller & Dymond 2015, Dymond et al. 2016, Basher et al. 2020) to determine annual contaminant loads and dynamic models such as eWater Source (Blyth 2018) and SWAT (Hoang 2019a, Hoang 2019b) for time series modelling. Mention must also be made of the development of the Auckland Council Fresh Water Development Tool in Auckland (Bambic & Riverson 2016, Bambic & Riverson 2017, Grant et al. 2018) and Physiographic modelling in the Northland Region (Rissmann et al. 2019); these tools are currently bespoke to the regions but could also be adapted to other regions.

Name	**Metrics**	**Datatype**	**Developer / owner**	**Description**	**Key reference**	**URL**	**Licence**
New Zealand Empirical Erosion Model (NZeem®)	Sediment load	Raster	MW / LCR	Mean annual erosion rates (tonnes of soil/km2/yr) in the North Island of New Zealand under current (2010) landcover	Dymond et al. (2010)	North Island: https://lris.scinfo.org.nz/layer/48178-nzeem-erosion-rates-north-island/ https://koordinates.com/from/lris.scinfo.org.nz/layer/48178/ South Island: https://lris.scinfo.org.nz/layer/48176-nzeem-erosion-rates-south-island/ https://koordinates.com/from/lris.scinfo.org.nz/layer/48176/	Landcare Data Use Licence
Streambank erosion susceptibility Index	Rank	Polygon shapefile	MfE MW / LCR	Predictions of the susceptibility to streambank erosion for every stream link in the RECv2.5 digital river network.	Smith et al. (2019)	https://data.mfe.govt.nz/layer/105771-streambank-erosion-susceptibility-index/ Mirror: https://koordinates.com/from/data.mfe.govt.nz/layer/105771/	Creative Commons Attribution 4.0 International
Estuary Trophic Index (ETI)	Trophic index	Text file	NIWA	Online tools for assessing the trophic state of estuaries in New Zealand.		Launch site: https://niwa.co.nz/freshwater-and-estuaries/research-projects/the-new-zealand-estuary-trophic-index Tool 1: Determining eutrophication susceptibility using physical and nutrient load data: https://shiny.niwa.co.nz/Estuaries-Screening-Tool-1/	

				Uses modelled and measured nutrient loads delivered to estuaries.		Estuary data, including modelled estimates of nutrient (CLUES) and sediment (NZSYE) loads delivered to estuaries, are available for download from site as a text file. Tool 2: Assessing Estuary Trophic State using measured trophic indicators: https://shiny.niwa.co.nz/Estuaries-Screening-Tool-2/ Tool 3: Assessing Estuary Trophic State using a Bayesian Belief Network: https://shiny.niwa.co.nz/Estuaries-Screening-Tool-3/	
River Water Quality[2] Modelled State	Various water quality metrics including nutrients, visual clarity and turbidity	Text file	MfE / NIWA	Outputs of Random Forest modelling provided for every reach in the REC 2.5 network	Whitehead (2019)	https://data.mfe.govt.nz/table/99871-river-water-quality-modelled-state-20132017/ Data can also be viewed as a map and downloaded from the NIWA River Maps tool: https://shiny.niwa.co.nz/nzrivermaps/	Creative Commons Attribution 4.0 International
CLUES	E. coli	Polyline shapefile	MfE / NIWA	National run of the CLUES E. coli model to support policy development for swimmable rivers	Semadeni-Davies et al. (2018)	Baseline scenario: https://data.mfe.govt.nz/layer/98358-water-quality-for-swimming-scenario-0/	Creative Commons Attribution 4.0 International
New Zealand Sediment Yield Estimator[3]	Sediment loads	Polyline shapefile	MfE / NIWA	Sediment loads estimated for every reach in the REC 2.5 network	Hicks et al. (2019)	https://data.mfe.govt.nz/layer/103686-updated-suspended-sediment-yield-estimator-and-estuarine-trap-efficiency-model-results-2019/	Creative Commons Attribution 4.0 International