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Using marine ecoengineering to mitigate biodiversity loss on modified structures in the Waitematā Harbour.

A thesis presented in partial fulfilment of the requirements for the
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Abstract

The construction of infrastructure on the foreshore is an unavoidable consequence of an ever-expanding human population. Traditionally, this infrastructure has replaced soft-substrates with hard substrates. Furthermore, even for native biota which occupy hard substrates, the flat, featureless construction of most marine infrastructure provides little habitat heterogeneity and results in depauperate communities with little biotic resistance against non-indigenous species. Marine ecoengineering provides a possible solution to this global phenomenon by using intelligent construction techniques that promote the accumulation of native biodiversity. Here, I used eco-engineered settlement plates to examine the effect of habitat complexity on the biodiversity of communities inhabiting existing. Additionally, we examined the effects of climate change driven increases in rainfall on the performance of ecoengineered substrates in the mid-intertidal zone. Last, we reviewed and synthesised the available literature on the species present in The Waitematā Harbour and, to the best of my knowledge, provide the most complete species lists to date.

In chapter two, we transplanted eco-engineered settlement plates seeded with local bivalve, *Perna canaliculus*, onto an existing seawall and monitored the accumulation of biodiversity. Overall, we show that both structural and biological habitat heterogeneity enhanced the biodiversity of the seawall community. Additionally, we found that the cemented pavement of volcanic rock that constituted the existing seawall, accumulated biodiversity faster than flat concrete settlement plates, supporting the use of this type of seawall construction over flat concrete seawalls. However, benefits to biodiversity could be further enhanced by explicitly adopting ecoengineering designs that provide crevices for intertidal organisms.

In chapter three, we examined the performance of ecoengineered substrates under the prediction that climate change will enhance rainfall by 20% in the Auckland region. While no effect of increased rainfall was observed for the mobile invertebrate community or the flat plates, increased rainfall did influence the biodiversity of the fouling community on the ridged plates, likely as a consequence of reduced desiccation stress. Although this was only a short-term experiment we predict that given time to develop, a distinct fouling community could influence the diversity mobile invertebrate community, shifting the whole community vertically up the seawall.

The review of the Waitematā taxonomy presented in chapter four, provides a reference for future studies of the biodiversity of the Waitematā harbour as well as identifying several gaps in our understanding, a cause for concern. Specifically, we show that non-indigenous species make up a considerable proportion of the fouling species listed for the Harbour and suggest that some of this could have been avoided by the adoption of ecoengineering techniques.

Overall, this thesis recognises that habitat heterogeneity, be it natural or man-made, is a vital driver of biodiversity. Each chapter provides additional insight, supporting the benefits of marine ecoengineering. These positive results within the Waitematā Harbour show potential for larger scale experimental trials and for the broader application of these techniques in other locations. By implementing intelligent design and eco-friendly materials in marine infrastructure, we can reduce the impact on local intertidal communities and indirectly reduce the spread of non-indigenous species.

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Overwhelming thanks,

Connor James M^{c}Kenzie

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Chapter 1 – General Introduction

1.1 Introduction

As a transition zone between the terrestrial and marine environments, the intertidal zone is a unique and harsh environment characterized by immense spatial and temporal environmental variation. Extending from the spring low tide line, which is nearly constantly inundated, to the spring high tide line, which is nearly constantly exposed, intertidal zones have great variation in their area, substrate, and complexity. Substrates of the intertidal zone include rugged cliff faces, sandy beaches, mudflat, saltmarshes, or wetlands.

The intertidal zone can maintain numerous microhabitats populated by distinct communities, the presence of these distinct communities, often with abrupt transitions, lead to the division of the intertidal zone into three vertically distinct zones. Originally labelled as the Littorina zone, Balanoid zone, and sublittoral fringe in early studies (Stephenson & Stephenson, 1949), these zones within the intertidal area have been continually revised over the years. Now, generally, the three zones are often colloquialised as the high, middle, and low tidal zones respectively, with the higher and lower zones represented by the tidal heights excluded during a neap tidal cycle.

With regular inundation and exposure, organisms that inhabit the intertidal zone must survive exceptional environmental extremes. Intertidal species must endure potential gradients from baking in the sun to freezing cold; desiccation; harsh waves that can dislodge inhabitants; as well as hyper- and hyposaline conditions during low tide. Yet, the intertidal zone supports unique communities adapted to such conditions and is utilised by a diverse range of fauna from both terrestrial and marine sources (Paine, 1994).

Some intertidal zones are vital for maintaining the lifecycle of unique, threatened, and/or commercially important species. Various seabirds and fish forage within intertidal zones, while mudflats and wetlands are utilised by the juvenile stages of numerous fish species, some of which support significant commercial or recreational fisheries (NIWA, 2007). Therefore, it is of ecological and financial benefit to protect the intertidal zone from potential stressors occurring from anthropogenic modification.

1.1.1 Marine anthropogenic interactions

The foreshore is the connection to the ocean for humans and has been utilised as a vital resource for sustenance, recreation, and waste removal, while facilitating trade over long distances by sea. As time and technology have progressed, coastal populations have swelled immensely and coastal urban environments now accommodate millions of humans (Neumann *et al.,* 2015). As the human population has grown, so too have anthropogenic effects on the foreshore. It is now a predictable consequence that wherever the urban sprawl meets the ocean, the foreshore is physically modified. Unless foreshores in proximity of urban populations are protected, seafood may be over harvested, diverse habitats are replaced by uniform modifications, vital breeding grounds are degraded or removed, and waste and storm water is discharged making entire sections of coastline unfit for recreation and harvesting (Bess, 2010).

1.1.1.1 Direct modification

To facilitate coastal living, humans have modified the foreshore often with little regard for the environmental effects of their actions. Modifications include structures such as wharves, marinas, and ramps, for sheltering and launching ships; larger structures such as ports and docks for the loading and unloading of cargo and people; as well as structure for coastal defence such as seawalls, breakwaters, and groynes. The intensity of modification often directly relates to the local population size, from small rural towns having a minimal effect, to metropolitan harbours serving as hubs of international shipping routes and housing thousands of recreational vessels.

In general, the foreshore surrounding coastal settlements is being converted into a ubiquitous vertical smooth marine concrete pavement. The implementation of artificial structures often results in a distinct assemblage establishing on the new hard substrate (Glasby, 1999). As spatial heterogeneity tends to enhance biodiversity, the uniform complexity of artificial structures allows fewer species to co-exist, lowering local biodiversity (Levin, 1992). Therefore, even in cases where natural hard substrates are replaced, the communities that re-establish are significantly less diverse and as a result, artificial coastlines cannot currently be considered as a viable substitute for a natural shoreline (Glasby & Connell, 1999; Chapman & Bulleri, 2003).

Seawalls lack features such as rockpools and shaded crevices that provide protection from desiccation, predation, and wave action, leaving the intertidal community constantly exposed to the elements (Firth *et al.,* 2013). This results in vast differences in the general abundance and community composition of species, and habitat used by mobile fauna

(Moreira *et al*., 2007). Without the habitat heterogeneity, the vertically distinct intertidal communities have been found to have a much greater overlap on seawall than a natural sloping coastline (Bulleri & Chapman, 2010). As a result, some species have developed interrelationships that have not been found on rocky shores (Fairweather, 1988), while some mobile invertebrates have a reduced body size and reproductive output in response to the additional community density (Moreira *et al*, 2006). Ultimately, urban marine infrastructure is not a surrogate for natural hard foreshore environments, since the provision of this habitat alters local biodiversity by modifying the frequency, dispersal, fitness, and behaviour of local communities, which consequently facilitates the establishment of non-indigenous species (NIS) into the community (Bulleri & Chapman, 2010).

1.1.1.2 Indirect Modification

Areas with direct marine modification are usually associated with substantial pollution from wastewater, anti-foul leaching, and oil spills from nearby marine traffic; which is highly detrimental to the fitness of the local native communities (Piola & Johnston, 2008). Comparative studies between hard-substrate communities on settlement plates around polluted seawalls and nearby natural reefs, found that in the heavily polluted areas, NIS significantly increased in richness and dominance on the plates (Piola & Johnston, 2008). An additional treatment of slow releasing heavy metal pollutant (Cu) led to a reduction in the native community's richness and dominance, independently from the environmental conditions (Piola & Johnston, 2008). However, heavy metals are not the only common form of anthropogenic pollution. Agricultural runoff including excess inorganic nutrients such as nitrogen and phosphorus into streams and rivers, accumulates within the water column and can cause devastating effects in estuaries and the ocean (Ryther & Dunstan, 1971).

The inorganic nutrients are essential for the growth of algae and macrophytes, which form the lowest trophic level of the aquatic food webs (Pace *et al.,* 1999). However, these primary producers utilise relatively low amounts of these nutrients. The vast excesses contributed from agricultural processes, exceeds the level that primary consumers can cope (Ryther & Dunstan, 1971). This process, known as nutrient overloading or anthropogenic eutrophication, results in harmless algal blooms; unless formed of cyanobacteria. Cyanobacteria produce cyanotoxins that poison ecological communities and accumulate in the flesh of fish and shellfish (Ibelings & Chorus, 2007). If infected seafood is consumed by humans, it can increase their chances of neurodegenerative diseases such Alzheimers, Parkinson's, and ALS (Holtcamp. 2012). However, it is the after effect of a short lived algal bloom that causes the most ecological devastation, as sudden collective decomposition by micro-organisms consumes all the oxygen suspended within water column. The lack of oxygen subsequently suffocates the local communities creating dead zones devoid of nearly all marine life (Joyce, 2010). By 2008, 405 oceanic dead zones had been recorded, the most significant (such as those within the Baltic Sea and Gulf of Mexico) covering tens of thousands of square kilometres (Diaz & Rosenberg, 2008).

1.1.1.3 Climate change

As one of the greatest modern threats to global ecological communities, anthropologically facilitated climate change has become a primary interest for research. The collective atmospheric pollution resulting from facets of industry, energy production, and livestock rearing, is worsening a greenhouse effect across the planet, raising temperatures at a rate significantly faster than the planet's natural climate oscillation (IPCC, 2014). The potential intensity and rate of climate change are constantly debated. However, various models and projections are indicating a general trend of increasing air and sea surface temperatures, ocean acidification, changes in precipitation patterns, increasing heatwaves and droughts, more powerful storm systems, and a reduction/loss of glaciers and polar ice caps (IPCC, 2014). The latter results in a rise in the current sea levels over the next few centuries and the accumulation of all effects radically modifying global ecological systems (Stocker, 2014).

In the ocean, climate change has already caused devastation on coral reef communities, as the absorption of atmospheric CO_2 results in acidification of the ocean and reduces the rate of calcification in reef-building corals. As consequence, large areas of coral reef are dying as they are unable to adapt to these sudden climatic shifts (Raven *et al.,* 2005It has been predicted that rising sea temperatures will support the propagation of marine NIS, increasing their dominance within fouling communities (Sorte *et al.,* 2010: Cockrell & Sorte, 2013). As one of the most well studied natural environments, the rocky intertidal zone is speculated to be the best natural laboratory in which to monitor the effects of subtle climate changes, occurring in both the terrestrial and aquatic environments (Helmuth *et al.,* 2006). This is because both terrestrial and marine stresses induced by climate change can have an observable effect on the vertical and geographic distribution of intertidal communities (Somero, 2002). With global warming a foremost concern of climate change, many publications focus on the potential effects from thermal variation. However, the intertidal zone could also be utilised to test the effects of more subtle or underappreciated climate change effects.

1.1.1.4 Marine non-indigenous species (NIS)

The cumulative effects of the anthropogenic modification are facilitating the spread of marine NIS across the globe. The conversion of coastlines into artificial hard substrate has allowed species that dominate in these uniform environments to propagate across the globe, capitalising on the ever-increasing international trade routes (Didham *et al.* 2007). NIS can survive lengthy journeys between nations fouled to the underside of vessels or within the ballast tanks of large ships, which alone are estimated to be transporting up to 10,000 species between biogeographical regions at any given moment (Carlton, J.T., 1999). On entering a new region, many NIS have difficulty settling within undisturbed native communities and therefore are usually unable to establish themselves within the environments that they travel to (Ferreira, 2003). Unfortunately, local species greatly suffer as a result of habitat modification and pollution, allowing NIS to more easily invade the affected areas, such as those found around hubs of transportation (Didham *et al.* 2007). This facilitates the establishment of the NIS into new bioregions by initially invading communities on artificial structures. With coastline modification occurring worldwide, NIS that are preadapted to these conditions, are initially invading international ports and marinas before exploiting the surrounding environments (Seebens *et al.* 2013).

1.1.1.4.1 Distribution

Any international vessel has potential to spread NIS into new geographical regions, and while commercial crafts are often blamed for their propagation, recreational vessels are also a significant factor both internationally and domestically (Davidson *et al.,* 2010). With international ports being key areas for the initial introduction of NIS to a region, domestic vessels can then spread species along the coastline to domestic ports and recreational marinas. A spatially explicit stochastic model incorporating ~1300 international and domestic yachts within New Zealand, simulated the spread of a hypothetical marine invader, and found that the 'hubs' that had greater recreational traffic developed a 75% greater chance of becoming infected than marinas with lower traffic (Floerl *et al*, 2009). This result shows that small domestic marinas also at high risk of invasion by NIS. With each hub infected, NIS have a greater potential to spread further along the coast and into the surrounding natural communities.

While marine NIS are invading new regions primarily through the facilitation of shipping networks, there are species which have been purposefully introduced for economic benefit that would have otherwise been unable to expand their range naturally. The earliest recorded cases of deliberate introductions are translocations of shellfish to new regions for harvesting. In the mid nineteenth century when this practice became common, there was little regard for the environmental impacts, with entire oyster bed communities being dredged and translocated across the North Sea. This consequentially introduced all the oyster bed associated species (Möbius, 1877: Korringa, 1976 a, b). While non-native shellfish are still being deliberately introduced to new areas, since the 1970s, the transplanted stock originate from enclosed hatcheries to ensure that only the desired species are introduced (Chew, 1990).

Farming of non-indigenous fish species has also contributed to the introduction of marine NIS into new bioregions. While most fish stocks currently farmed are fresh water species, mariculture is an expanding industry where mismanagement or equipment malfunctions can lead to large exotic species escaping into novel marine environments. Fish species utilised in mariculture practices are typically selected for their rapid growth and reproduction, (ICLARM, 1984) and are typically anadromous or able to tolerate wide salinity ranges (Payne, 1983). Therefore, farmed marine fish that escape from their facility have a strong disposition to naturalise within estuarine and brackish waters (Jaafar *et al*., 2012).

The most recent introduction pathway identified was the exotic fish trade, which has been acknowledged as a route for introduction after fish have been accidently or deliberately released by their owner (Whitfield *et al.,* 2002: Semmens *et al.,* 2004). In Singapore, the ornamental fish trade has become a major industry with an export value of 100 million Singapore dollars (Ministry of National Development, 2009). This has, however, resulted in several ornamental fish species becoming established within Singapore's estuarine communities (Jaafar *et al*., 2012), with one species of Poeciliidae becoming their most prominent species in brackish canals and mangrove streams (Lim & Low, 1998).

While most marine NIS will be limited to areas where there is a reduction of the native species from of the detrimental effects of direct and indirect artificial modification, it is possible for some NIS to establish within the natural communities or in areas of limited modification (Ferreira, 2003; Dafforn *et al.,* 2012). While pollution is usually associated to area of heavy anthropologic activity, this is not the case for artificial substrates, which can be found haphazardly throughout relatively pristine areas (Dafforn *et al.,* 2012). As a result, some species can spread far outside invasion hubs using modified structures e.g. groins, breakwaters, private wharves / piers, buoys, wreaked / inactive ships, or washed out debris, as stepping stones down the coastline to establish on (Glasby *et al*., 2007).

Fortunately, comparative studies have found that despite marine modification are abundantly colonised by NIS, it is rare for fouling NIS to establish into nearby undisturbed environments (Simkanin *et al*., 2013). This trend has also been seen with mobile NIS, with invasive fish species found to rapidly decline in abundance with distance away from the port (Stuart-Smith *et al.,* 2015). This supports that the propagation of NIS is primarily facilitated by the circumglobal modification of urban coasts, which preadapted species have been able to utilise.

1.1.1.4.2 Adaptations

The characteristics that allow particular marine species to become invasive have been considered from various approaches of their biology. Environmental tolerance appears to be a fundamental characteristic to the success of marine NIS. With a generally greater resistance to heavy metal pollution than native biota (Hall *et al.,* 1998), NIS can dominate polluted harbour environments over the local community. Hewitt and Hays (2002), found that most marine NIS have thermal tolerances that extend well beyond the extremes recorded in their original environment (Lewis *et al.,* 2006). NIS fouled on ship hulls have also been recorded as having significant tolerance of fresh water immersion, surviving the Panama Canal crossing, which typically lasts from 8-10 hours (Chapman *et al.,* 2013; Ros *et al.,* 2014).

A comparative study of hydroid assemblages between seawalls and natural reefs in the Mediterranean Sea found that seawall hydrozoans typically had limited investment in somatic tissue, short life spans (annual or sub-annual), free swimming medusa stages, and a proportionally higher reproductive effort when compared to local natural communities (Megina *et al*., 2013). Despite focusing on one class of organisms, this study highlighted how typical seawall colonisers and correspondingly NIS, are R-strategists, focusing on rapid reproduction and wide dispersal, perfect for colonising disturbed communities (Platt & Connell, 2003). With tolerance to unsavoury conditions being heritable, this R-strategy is believed to allow NIS to have adapted much quicker to unsavoury conditions, such as heavy metal pollutants, through a faster natural selection process (McKenzie *et al.* 2011).

Fouling NIS have also been found to be preadapted to anthropogenic dispersal. Comparing the attachment strength and drag coefficient of NIS to similar native fouling species, it was found that NIS had simultaneously a stronger attachment and lower drag co-efficiency than the native species (Murray *et al.,* 2009). This adaptation therefore minimises the chance of being dislodged during international travel.

1.1.2 Marine Ecoengineering

Marine modification in urban areas is inevitable. However, it is possible to mitigate the detrimental effects by employing engineering techniques that are more harmonious with the natural environment by reintroducing spatial heterogeneity to modified foreshores (Mayer-Pinto *et al.,* 2017). The concept of engineering, while sustaining the environment is colloquially known as green- or ecoengineering and was originally described as “the design of sustainable ecosystems [which] intends to integrate human society with its natural environment for the benefit of both" (Mitsch & Jorgensen, 1989). Ecoengineering has more commonly been utilised terrestrially to create green buildings, which have reduced waste, power and water use, are constructed out of materials that are recycled or cause minimal in environmental damage, but more importantly are designed to support a natural community on their exterior; reducing the contrast of a barren cityscape to that of the natural world (Bergen et al., 2001). With the success of terrestrial ecoengineering practices, recent research has recognised the benefits of utilising ecoengineering on marine infrastructure to maintain the biodiversity of the local fouling communities and consequentially reduce the propagation of NIS.

The common approach prevalent in current literature is to reintroduce habitat heterogeneity by creating artificial rock pools along the seawall and indenting depressions or crevices within the seawalls, where communities can find protection from the elements (Firth *et al.,* 2013). The addition of rockpools to seawalls has been found to increase the diversity of foliose algae, fouling sessile invertebrates, and mobile species with more significant effects at higher tidal heights, while expanding the distribution of species, generally confined to low shore levels, into rockpools outside their natural tidal range (Chapman & Blockley, 2009). Positive associations between complexity and biodiversity have previously been confounded by the assumption that increasing the habitable area will consequentially increase biodiversity. However, using settlement plates with equal areas but diverse designs, it was found that irregularly sized indentations benefitted marine fouling communities the greatest (Loke & Todd, 2016). Modifying marine construction practices to incorporate ecologically friendly designs, would reduce the loss of local biodiversity on artificial structures and reducing the spread of NIS through metropolitan environments (Mayer-Pinto *et al.,* 2017).

1.1.2.1 The World Harbours Project

Instigated by the Sydney Institute of Marine Science (SIMS), the World Harbours Project's (WHP) goal is to develop resilient urban ports and harbours through a global network of collaborating scientists. The WHP involved the participation of 16 countries monitoring the conditions of 25 international ports (http://www.worldharbourproject.org/). To achieve this goal, the WHP began by focusing on four common concerns shared by global harbours: monitoring the harbours' water and sediment quality; navigating issues surrounding the multiple use and users of harbours; ecoengineering of modified structures; and public education and outreach. The experiments described within this thesis were a collaboration with the WHP as Auckland's representative of the ecoengineering working group. For the ecoengineering workgroup, a replicate experiment was deployed within each partner harbour to allow a global comparative study. The experiment involved installing ridged settlement plates with three degrees of complexity onto seawalls within each harbour, and adding an additional aspect of complexity through biological structures by seeding half the treatments with a native bivalve. In addition to the designed complexity of the settlement plates, the concrete used for their construction was an eco-friendly blend made to be less acidic than regular concrete and more porous, further increasing fine-scale complexity to increase the diversity of microbiota (Thompson *et al.* 1996).

1.1.3 Waitematā Harbour

Despite Auckland city being surrounded by two harbours connecting to different seas, the smaller northern Waitematā Harbour (36^{O}50’ S, 174^{O}45’ E) is often referred to as the Auckland Harbour. With deeper waters closer to shore than the Manukau Harbour and connecting to the generally calmer Pacific Ocean on the eastern coast of northern New Zealand, the Waitematā Harbour is Auckland’s main ocean access to international shipping routes. At over 180 square kilometres (Waterhouse, 1966), the Waitematā Harbour was a key anchorage site for the indigenous Māori people and later European colonisers because of its lack of shoals and sandbars, while being well protected from Pacific storms by Rangitoto, Motutapu, and Waiheke Islands outside the harbour mouth, as well as the Hauraki Gulf itself (Hayward, 1983). Formed from a drowned valley, the Waitematā is commonly broken into two sections – the Upper Harbour entailing areas west of the Harbour Bridge, and the Mid Harbour containing the areas east of the Harbour Bridge to the harbour entrance between North Head and Bastion Point (Hounsell, 1935).

The Upper Harbour is defined by several estuarine rivers which feed into the harbour from an urban catchment in the south and a larger rural catchment in the north and west which consists predominantly of tidal mudflats and saltmarsh (Hewitt *et al.,* 2006). Although the Upper Harbour suffers from minimal direct anthropogenic modification it is susceptible to pollution from storm water discharge and nutrient overloading from agriculture runoff (Hewitt *et al.,* 2006). There are growing concerns that urban intensification over the next 50-100 years will increase storm water related contaminants to a level which will negatively affect the ecological functioning of the Upper Waitematā (Cummings *et al*. 2002).

The Mid Harbour has a southern coastline nearly completely modified and contains the Ports of Auckland, New Zealand’s largest marina in Westhaven, and Auckland’s CBD waterfront (Hounsell, 1935). The few sections without direct modification, such the sandy beach of Okahu Bay, are subject to such pollution that they have been restricted to the public for swimming and harvesting (Faaui, 2012). Beyond the harbour entrance lie several islands, which protect the harbour from significant storm swells and create a series of channels running away from the harbour, which has additionally been referred to as the lower or outer harbour by researchers (Dromgoole & Foster, 1983: Morley & Hayward, 2007).

With New Zealand's European colonisation beginning just over 200 years ago, the modification of the Waitematā Harbour was first recorded around 1860, when shores near the Auckland current CBD were filled and extended as wharves and breakwaters (Dromgoole & Foster, 1983). With the current population nearing 1.5 million people, the Ports of Auckland are New Zealand's largest international port with shipping links to 207 foreign ports in 73 countries by the late nineties (www.arc.govt.nz). With New Zealand having a much later introduction to international seafaring than other developed countries and because of the significant distance required to travel to it, there is potential that the marine ecosystem is less degraded by NIS. However, as a country which heavily relies on international shipping for over 95% of its trade in commodities (Inglis, 2001), the Ports of Auckland has not been exempt from bio-invasions, with 13 NIS recorded (Inglis *et al*., 2006).

The earliest taxonomic accounts of New Zealand's marine biota are thought to have been collected from around Auckland including the Waitematā. However, the locality of the specimens was never specified (Dromgoole & Foster, 1983). The first literature directly identifying Waitematā species was from Oliver (1923), who examined intertidal communities at Westmere Reef as an example in his wider account of New Zealand communities. In 1937, Powell performed the first large scale survey of benthic communities in the Waitematā and as the city has developed around the Waitematā, the local council has taken a greater interest in the ecological status of the estuarine system. A six-year ecological survey was undertaken from October 2000 to February 2006 identifying 92 confirmed NIS (Halliday *et al*., 2006). Overall, despite numerous biological surveys and studies on individual taxa, the literature on the biota of the Waitematā Harbour is scattered and much of the taxonomy requires updating and confirmation.

1.2 Thesis outline

This thesis is composed of an introduction (Chapter one), two research chapters (Chapters two & three), one literature review (Chapter four), and a general discussion (Chapter five). The manuscript chapters (2-4) contribute to our understanding of marine biota living within Auckland's Waitematā Harbour with additional focus on artificial seawall communities. With both research chapters investigating settlement plates on seawalls, there is some unavoidable repetition of the conceptual framework and methodological details among chapters two and three. Outlines for the subsequent chapters are as follows:

Chapter 2: Assesses the effects of marine ecoengineering to mitigate the loss of biodiversity and propagation of non-indigenous species resultant from marine modification. In addition to structural complexity, the effect of biotic complexity was assessed by seeding plates with a local bivalve *Perna canaliculus*. Relationships between the forms of spatial heterogeneity are discussed and environmental factors influencing the results are evaluated. Overall various analyses found a consensus that spatial heterogeneity on seawalls increases the biodiversity of fouling and mobile invertebrate communities.

Chapter 3: Addresses the potential consequences of climate change on seawall biodiversity, as a result from potentially increasing rainfall in Auckland. The influence of habitat complexity is reaffirmed and discussed in relation to the potential rainfall effect. A potential relationship of rainfall and diversity is discussed and the latent ramifications of such an effect on global intertidal communities is expanded on.

Chapter 4: Reviews the species found within the Waitematā Harbour, summarising them into an identification list, sorted by phylum, order, and family, crediting the earliest identification and the publications author(s). This review mostly covers publicly accessible research and data, however there was limited access to corporate data during the research. The abundance and nature of NIS within the Waitematā is discussed and implicated as a result of anthropogenic practices. While this list can still be expanded, on publication it will be the most complete account of the breadth of the Waitematā estuary biota.

Chapter 5: Reviews the previous chapters to underscore the key results of their findings and implicate the relevance of the findings on the status of metropolitan harbours.

Chapter 2: Ecoengineering enhances native species biodiversity on seawalls in the Waitematā Harbour.

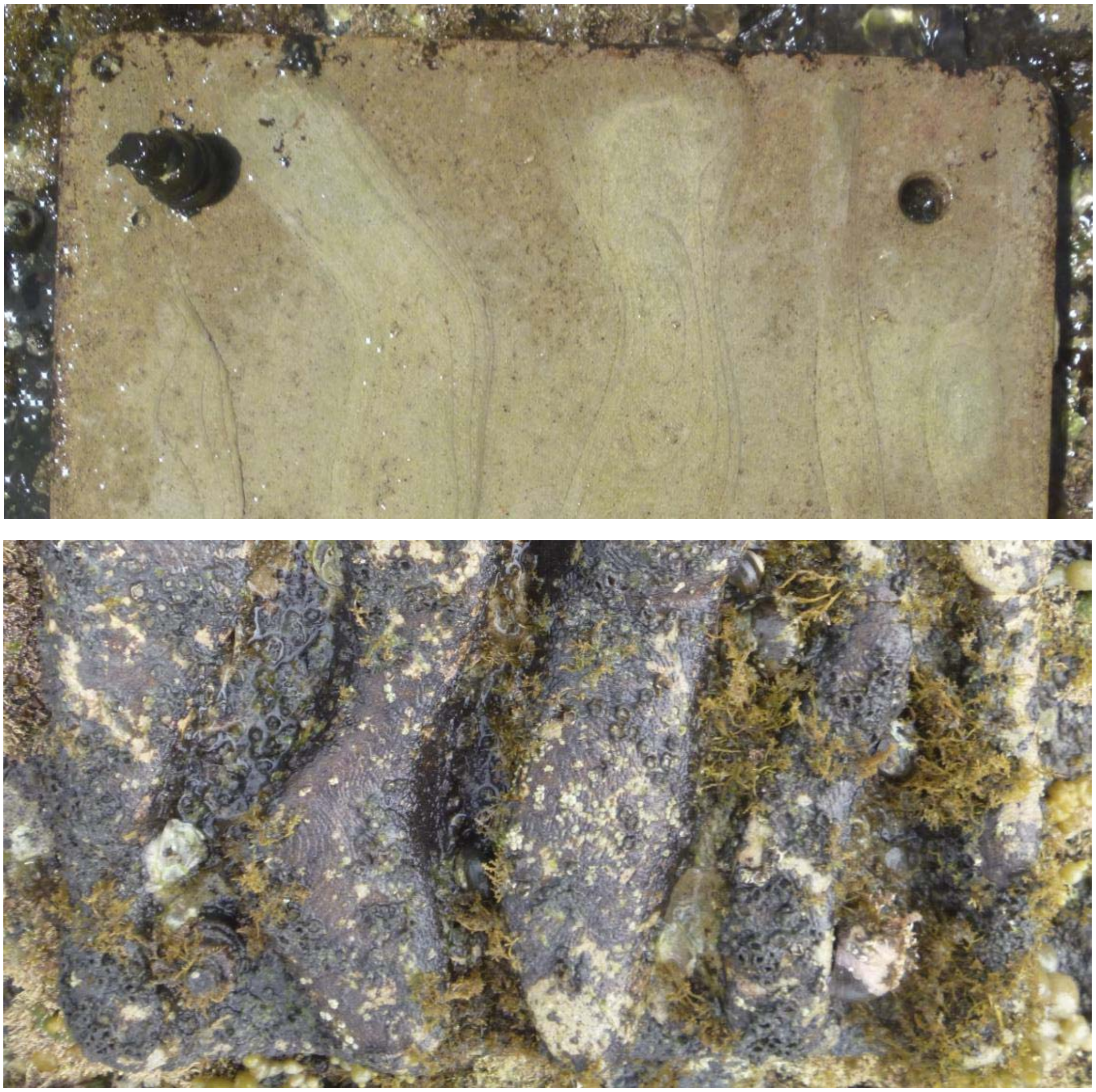

Photo: 2.1: Comparative photo combination of unseeded plates after 12 months at Westhaven. Top half - flat plate, bottom half - 2.5cm ridged plate. Photographer: Connor M^{c}Kenzie.

2.1 Abstract

Ecoengineering has been proposed as a solution to the common situation where communities inhabiting artificial marine infrastructure are typically less biodiverse and have a higher proportion of non-indigenous species (NIS) than neighbouring rocky shore communities on unmodified substrates. Here, we quantified the effects of using eco-engineered substrates that enhance structural complexity as an alternative to traditional seawall construction materials at two sites in the Waitematā harbour. In addition to structural complexity, the effects of biotic complexity were assessed by seeding plates with a native bivalve, *Perna canaliculus*. We found a consistent positive effect of increased habitat heterogeneity on biodiversity. In general, both structural complexity and biological complexity enhanced biodiversity, with many species preferring the crevices provided by each factor. Overall, we found that habitat heterogeneity is crucial for maintaining biodiversity, and the implementation of intelligently designed artificial structures could mitigate the global degradation in urban marine environments.

2.2 Introduction

2.2.1 Coastal homogeneity

Spatial heterogeneity is a ubiquitous driver of biological diversity (Smith, 1972). While often confounded with an increase of habitable area, the complexity of a habitat regulates the availability of discrete microhabitats and niches, allowing biodiversity to increase independent from the total area (Kostylev *et al.,* 2005). Heterogeneity of the environment provides resource variability, allowing species that could not co-exist within a homogenous environment to co-exist (Levin, 1992). The correlation between complexity and biodiversity has been observed across all ecosystems, emphasised by rainforests and coral reefs, with the greatest biodiversity on land and sea respectively (Tews *et al.,* 2004). It can even be observed at a micro-scale, with microbiota communities being more diverse on rough surfaces than smooth ones (McCormick, 1994: Thompson *et al.,* 1996). However, anthropogenic modification tends to homogenise natural environments, with drastic negative consequences for biodiversity.

An estimated 60% of the human population live within 100km of the ocean (Cohen *et al.,* 1997) and foreshores across the globe have undergone significant modification as a result. Land reclamation and artificial structures reduce the spatial heterogeneity of coastlines by converting the existing intertidal zones into uniform simplified coastlines. The loss of complexity does not allow diverse communities to re-establish, due to the greater competition over the fewer resources which remain (Levin, 1992). Pollution from marine vessels, storm and waste water releases toxins into the marine environment, which are absorbed into the tissues of the primary producers and can bioaccumulate in higher trophic levels (Sivonen & Jones, 1999). Increased sedimentation from land use change also degrades the local foreshore by enhancing turbidity, reducing photosynthesis, and smothering the benthos (Wolanski & Spagnol, 2000). In severe cases, nutrient overloading from agriculture or industrial discharge can create ‘dead zones’ where nearly no marine life can survive, the worst covering thousands of square kilometres (Diaz & Rosenberg, 2008). These compounding negative effects can significantly reduce biodiversity around urbanised coastlines.

Marine modifications are synonymous with coastal human populations, with the construction of piers, boat ramps, seawalls, breakwaters, groynes, marinas, and docks. The amount of modification often correlates with the population size, with minimal disturbance around small towns, while dense urban areas are subjected to heavy modification (Williams & Thom, 2001). These new substrates are not alike to natural rocky reefs as they lack any features such as rockpools, crevices, or overhanging ledges which provide protection from desiccation, predation, and wave action. Without the habitat heterogeneity provided within natural intertidal zones, vertically distinct communities have a greater overlap on seawalls, than a natural, sloping coastline (Bulleri & Chapman, 2010). As a result, some species have developed interrelationships that have not been found on rocky shores (Fairweather, 1988) and some mobile invertebrates have a reduced body size and reproductive output in response to the amplified community density (Moreira *et al*, 2006). While it is expected that introducing a novel substrate into the marine environment would attract a distinct community, when compared to communities on rocky reefs, the artificial substrate’s communities have an overall reduced biodiversity (Glasby, 1999).

Many non-indigenous species (NIS) initially settle into anthropogenically modified structures, where the diversity of native species is often reduced (Bulleri & Chapman, 2010). In marine systems, the epicentres of international transport are the leading vectors for the introduction of NIS. The concomitant effects of supply and reduced local biodiversity facilitate the proliferation of NIS across the globe. Therefore, modified coastlines are at risk of losing their unique local communities and becoming homogenous globally.

2.2.2 Ecoengineering

While marine modification in urban areas is inevitable, it is possible to mitigate detrimental effects on the local community by employing engineering techniques which are more harmonious with the natural environment and reintroduce spatial heterogeneity to the modified foreshore. The concept of engineering while sustaining the environment is colloquially known as green- or ecoengineering and was originally described as “the design of sustainable ecosystems [which] intends to integrate human society with its natural environment for the benefit of both" (Mitsch & Jorgensen, 1989). Ecoengineering has more commonly been utilised terrestrially to create green buildings, which have reduced waste, power and water use, are constructed out of materials which are recycled or have minimal in environmental damage, but more importantly are designed to support a natural community on their exterior; reducing the contrast of a barren cityscape to that of the natural world (Bergen *et al.,* 2001). With the success of terrestrial ecoengineering practices, recent research has recognised the benefits of utilising ecoengineering on marine infrastructure to maintain the biodiversity of the local fouling communities and mitigate the propagation of NIS (Chapman & Blockley. 2009).

The earliest forms of marine ecoengineering were man-made ‘reefs’ created through the deliberate sinking of decommissioned vessel in areas where natural reefs are absent. While the new artificial reefs are more diverse than the previous benthos, at a finer scale, the spatial complexity of such vessels is minimal and thus these reefs tend to capture only a subset of the biodiversity of a naturally inhabiting rocky reef (Bumbeer & Rocha, 2012). As the novelty of the structures attract recreational vessels for diving and fishing, they usually have greater connectivity with urban harbours than natural rocky reefs. Therefore, while these reefs may boost the biodiversity of the immediate area, they can act as reservoirs for invasive species far from urban development where there is the potential for active management (Bumbeer & Rocha, 2012). These early examples highlight how artificial, spatial heterogeneity can influence biodiversity in the marine environment, while emphasising that a finer scale of complexity should not be ignored.

While spatial heterogeneity is beneficial to biodiversity, not all forms of complexity are equally beneficial to a community and the optimal complexity type can change depending the community under consideration. Within a rocky intertidal foreshore, indentations in the substrate have been identified as the optimal level of heterogeneity to enhance biodiversity (Beck, 2000: Loke & Todd, 2016). Attributes such as pocks, crevices, overhangs, and rockpools provide organisms with shade and water retention, relieving thermal and desiccation stress, and protection from predation and braking waves. Preliminary intertidal ecoengineering studies have thus focused on reintroducing these features to seawalls to try to increase biodiversity. Artificial rockpools attached to seawalls at various tidal heights increased diversity of foliose algae, sessile foulers, and mobile species, with the greatest effect at higher tidal heights because it allowed vertical expansion of species previously confined to the low intertidal (Chapman & Blockley, 2009).

The development of concrete mixes with a reduced pH allow the creation of marine structures that are more aligned to the oceans natural pH than traditional structures (Rui, 2014). These eco-concretes are also more porous than the traditional concrete, allowing for greater fine-scale heterogeneity. A study of intertidal microbiota found that even micro-scale spatial heterogeneity enhanced the diversity of microbiota occurring on rough surfaces relative to those on smooth rock (Thompson *et al.,* 1996). Therefore, if marine modifications utilise these concretes in combination with intelligent, spatially complex designs, the decline in local fouling biodiversity on artificial seawalls can be minimised.

2.2.3 The World Harbours Project

Instigated by the Sydney Institute of Marine Science (SIMS) in 2014, the World Harbours Project (WHP) set the goal to develop resilient urban ports and harbours through globally integrated research and management (http://www.worldharbourproject.org/). The WHP involves the participation of 16 countries, monitoring the conditions of 25 metropolitan harbours. There are four major working groups within the WHP: water and sediment quality; ecoengineering; navigating the competing interests of multiple users; and public education and outreach. For the ecoengineering working group, the SIMS and collaborating institutes designed an experiment which was applied by each of the partner's home harbour. This experiment involved deploying spatially heterogeneous settlement plates, which had ridges and crevices of varying height onto the seawalls within each harbour. In addition to this physical heterogeneity, the effects of biological heterogeneity were explored by seeding the plates with mature native bivalves. The settlement plates were designed and manufactured by Reef Design Lab, who use a marine friendly concrete blend to produce settlement plates which are heterogeneous for both macro- and micro-fauna.

This study was New Zealand's contribution to the WHP ecoengineering working group. The settlement plates were deployed in the Waitematā Harbour, which is the primary ocean access for New Zealand's largest city, Auckland. The Waitematā harbour is also the most heavily modified marine environment in New Zealand. The Waitematā Harbour contains the country's largest shipping port, marina, and naval base, while also supporting two additional ship ports, six additional marinas, nine ferry terminals, and is scattered with wharves, buoys, moorings, and seawalls across its coastline. The two seawalls selected for the deployment of the settlement plates were within the heavily modified mid-Harbour area of the Waitematā, one near the Devonport Naval Base and the other near New Zealand's largest marina at Westhaven.

2.3 Methods

The sampling methods used in this survey were based on protocols from the Sydney Institute of Marine Science for the World Harbours Project. In each harbour, two seawall locations were selected for the attachment of thirty 25x25cm concrete settlement plates with three levels of complexity: flat, 2.5cm ridges, 5cm ridges. The settlement plates were designed and manufactured by Reef Design Lab who employ a marine friendly concrete blend. Half of the plate treatments had an equal number of a native bivalves attached (enough to cover 35% of the flat plate) with a harmless epoxy; making six treatment types with five replicates at each seawall. An additional five treatment of 25x25cm natural seawall, scrapped clean of previous fouling, was included as a control. For Auckland, the green lipped mussel (*Perna canaliculus*) was selected as the native bivalve, as it is the most abundant indigenous bivalve on its seawalls.

2.3.1 Bivalve preparation

By adhering mussels on the plates in groups of four, the optimal size of *P. canaliculus* that allowed 16 individuals to cover 35% of a flat plate was calculated, roughly 65mm long and 35mm wide. An initial collection of 20 mussels from a local population at Northhead, Auckland (36 °49’32”S, 174°48’47”E) was cleaned of epibionts and placed alive within tanks of salt water that were changed every second day. These mussels were glued to either a smooth pavers or volcanic rock (typically used for seawalls in Auckland), using two-part Splash Zone Compound, Underwater Epoxy Putty, A-788. They were adhered using a ball of epoxy ~2cm in diameter, which was pressed onto the side of mussel furthest from where its two valves meet (to prevent gluing them shut). The epoxied side was then immediately pressed onto the hard surface. After one month, these trials were deemed successful as none of the mussels had detached or died. A further 480 mussels were collected from the same local population, which were cleaned of fouling before being glued in groups of 16 onto ten settlement plates of each complexity type using the Splash Zone epoxy. On the ridged plates, the mussels were divided between the crevices and ridges, totalling eight each. The experimental plates, including the non-seeded treatments, were kept in tanks of saltwater for two weeks while the seawalls were prepared for their attachment. The water within the tanks was changed every two days, collected from the local Browns Bay Beach or from either of the seawalls locations

2.3.2 Site preparation and deployment

At each seawall, thirty-five 25x25cm plots were scrapped and wire brushed to remove existing fouling at intertidal height (0.8m above chart datum), where *P. canaliculus* naturally occurred, no less than one meter apart along the seawall. Two holes, 8mm in diameter, 70mm deep were drilled in opposite corners of each scrapped plot on the seawall for attachment of the settlement plates. The plates were deployed into the field randomly over four days, with approximately half the plates being deployed at each site per day. Each day, the treatments assigned to the respective half of the seawall were loaded into plastic storage containers filled with saltwater and taken to the field site. The plates were then attached with the ridges in a vertical orientation using two stainless steel dynabolts (8mm x 80mm), with an 8mm nylon and stainless-steel split washer between the plate and the dynabolt nut. Two-part Splash Zone Compound, Underwater Epoxy Putty, A-788 was used as an additional adhesive behind and around some edges of the plates.

Half the mussels on each seeded plate were randomly selected, divided evenly among the crevices and ledges on the complex plates. These mussels were assigned a number and their locations were recorded on a template for each seeded plate. The mussels had their original height and length (Figure 2.1) recorded using callipers. However, the crevices restricted the access of the callipers and therefore only the bivalve height was recorded within them.

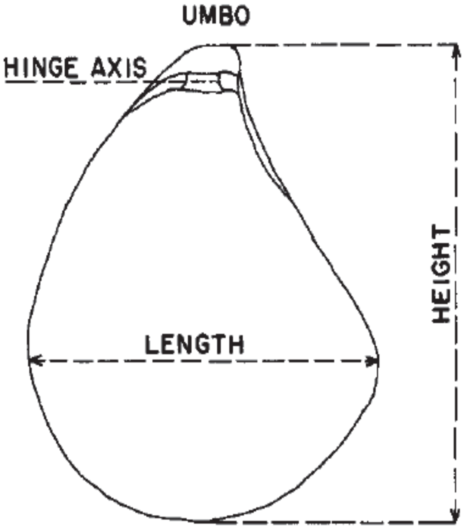

Figure 2.1: Reference for bivalve measuring.

2.3.3 Settlement plate monitoring

Observations of the settlement plates were performed after months 1, 3, 6, 9, and 12. Observation could only occur during spring low tides and required 2-3 low tides per site, depending on weather conditions and the minimum height of each tide. During these observations, each plate and scrapped section of seawall had three main attributes monitored: Fouling coverage through photo analysis, identification and abundance of mobile invertebrates, and health and growth of the seeded mussels where applicable.

For each plate, a photo was taken 0.5m from the front of the surface with two replications to ensure that I obtained an in-focus picture covering the entire plate area which would be used during later photo analysis. Several photos of each plate would then be taken at various perspectives with focus on highly diverse areas, mobile invertebrates and uncommon / rare species. The photo analysis was performed using Coral Point Count with Excel extensions program, which overlaid the area of the plates with random points. On each plate, 100 points were used to identify species presence and estimate the coverage of the fouling community. On the complex plates, 50 random points each were used from the areas on the ridge and crevices.

All mobile invertebrates on a plate were identified, counted, and recorded. Any new species found that were not immediately identifiable were assigned an operational taxonomic unit until their photos could be referred to experts for identification. Three areas of each plate were ignored for physical and photo identification: on the bolt ends, hexnuts, and washers; within the two remaining empty bolt holes; and the first 2cm of the base of each plate (i.e. the base of the flat plates).

On plates with seeded mussels additional observations were recorded regarding mussel health and size. The survival of the bivalves was recorded for each plate and classified as into 6 categories: alive, dead - valves intact, dead – single valve, dead – drilled, dead – cracked, missing. Mussels with both valves intact but no visible sign of damage were classified as valves intact, however if the top valve showed evidence of drilling or was crack they were classified respectively. In most cases, the top valve detached after the death of the mussel, with the remaining valve adhered to the plate, therefore it was not possible to identify the cause of death and was classified as a single valve. Mussels that had their original measurements recorded, were remeasured with the same callipers if they were still alive during each observation. These bivalve measurements were recorded for compliance with the WHPs protocols and were not analysis in this thesis.

2.3.4 Seawall deconstruction

After the 12-month observation period, the seawalls were returned to their original condition. The plates were then carefully removed from the seawall and had their back scoured to remove any fouling that had accumulated there. Each plate was then placed into a labelled plastic bag and carefully packed into plastic storage containers for transport back to the laboratory where they were frozen until the final destructive sampling. Only the drilled holes remained within the seawall.

2.3.5 Destructive sampling

Each plate was individually removed from the freezer and thawed for an hour before sampling. For seeded plates, the number of live and seeded bivalves were recorded and each mussel which survived had its height, length, and width recorded with callipers. All fouling species were identified on each plate and the areas which they covered on each habitat were estimated by eye. Conspicuous mobile species were collected from the plates, identified, enumerated. The ridges and crevices were then flushed individually over a 500um sieve and any remaining invertebrates found were recorded and added to their respective collections.

2.3.6 Analysis

Our analysis considered enhancement as a fixed effect with three levels: flat, 2.5cm ridges and 5cm ridges. Seeding was considered as a fixed effect with two levels: seeded and unseeded. Lastly, months were considered a fixed factor effect with four levels: 1, 3, 6 and 12 months. Location was considered a random effect with two levels: Devonport and Westhaven. All univariate analyses were conducted in R-studio, though the correct denominator for the F-ratio and the denominator degrees of freedom were adjusted manually given the random effects in our models. All multivariate analyses were conducted in PERMANOVA+ (Anderson *et al.,* 2005). Significance was quantified with a p-value less than 0.05.

2.4 Results

Non-indigenous species were rare in our experiment, apart from the naturalised *Crassostrea gigas*, of the 27-fouling species confirmed during the observation period, only one occurrence of an invasive species was identified (*Watersipora* sp.). None of the mobile invertebrates identified were confirmed to be invasive during this period.

At Westhaven, the seeded mussel's population deployed on the settlement plates was functionally extinct by the ninth month. At Devonport, a quarter of the seeded mussels survived the 12-months on the seawalls.

Table 2.1: Significance of complexity types on fouling community biodiversity at highest interaction (NS = no significance, +VE or -VE = significant positive or negative effect, respectively, † = marginal non-significance, M = significant treatment by Month interaction, L = significant treatment by Location interaction, L:M = significant treatment by location and month interaction)

Spatial heterogeneity	Cover (%)	Richness	Simpsons	Shannons	Jaccards	Bray-Curtis
Enhancement	L	+VE	L	L : M	L	L
Seeding	L : M	+VE	NS	NS	NS	L
Enhancement:Seeding	NS	NS	L	NS	+VE	+VE
Habitat utilisation						
Habitat	NS	†	†	+VE	N/A	N/A
Seeding	L	NS	NS	NS	N/A	N/A
Enhancement	NS	NS	NS	NS	N/A	N/A
Habitat:Seeding	NS	NS	NS	NS	N/A	N/A
Enhancement:Habitat	NS	NS	NS	NS	N/A	N/A
Enhancement:Seeding	NS	NS	NS	NS	N/A	N/A
Enhancement:Seeding: Habitat	NS	NS	NS	NS	N/A	N/A
Seawall control						
Enhancement	NS	+VE	L	L : M	M : L	M, L

Table 2.2: Significance of complexity types on mobile community biodiversity at highest interaction (NS = no significance, +VE or -VE = significant positive or negative effect, respectively, † = marginal non-significance, M = significant treatment by Month interaction, L = significant treatment by Location interaction, L:M = significant treatment by location and month interaction)

Spatial heterogeneity	Abundance	Richness	Simpsons	Shannons	Jaccards	Bray-Curtis
Enhancement	L : M	L	M	L	M : L	M : L
Seeding	L, M	+VE	M	-VE	+VE	L
Enhancement:Seeding	L	†	NS	-VE	+VE	+VE
Habitat utilisation						
Habitat	+VE	+VE	NS	+VE	N/A	N/A
Seeding	+VE	+VE	NS	NS	N/A	N/A
Enhancement	NS	L	NS	NS	N/A	N/A
Habitat:Seeding	NS	NS	NS	+VE	N/A	N/A
Enhancement:Habitat	NS	NS	NS	+VE	N/A	N/A
Enhancement:Seeding	NS	NS	+VE	NS	N/A	N/A
Enhancement:Seeding: Habitat	NS	NS	NS	NS	N/A	N/A
Seawall control						
Enhancement	NS	+VE	L, M	L : M	M : L	M : L

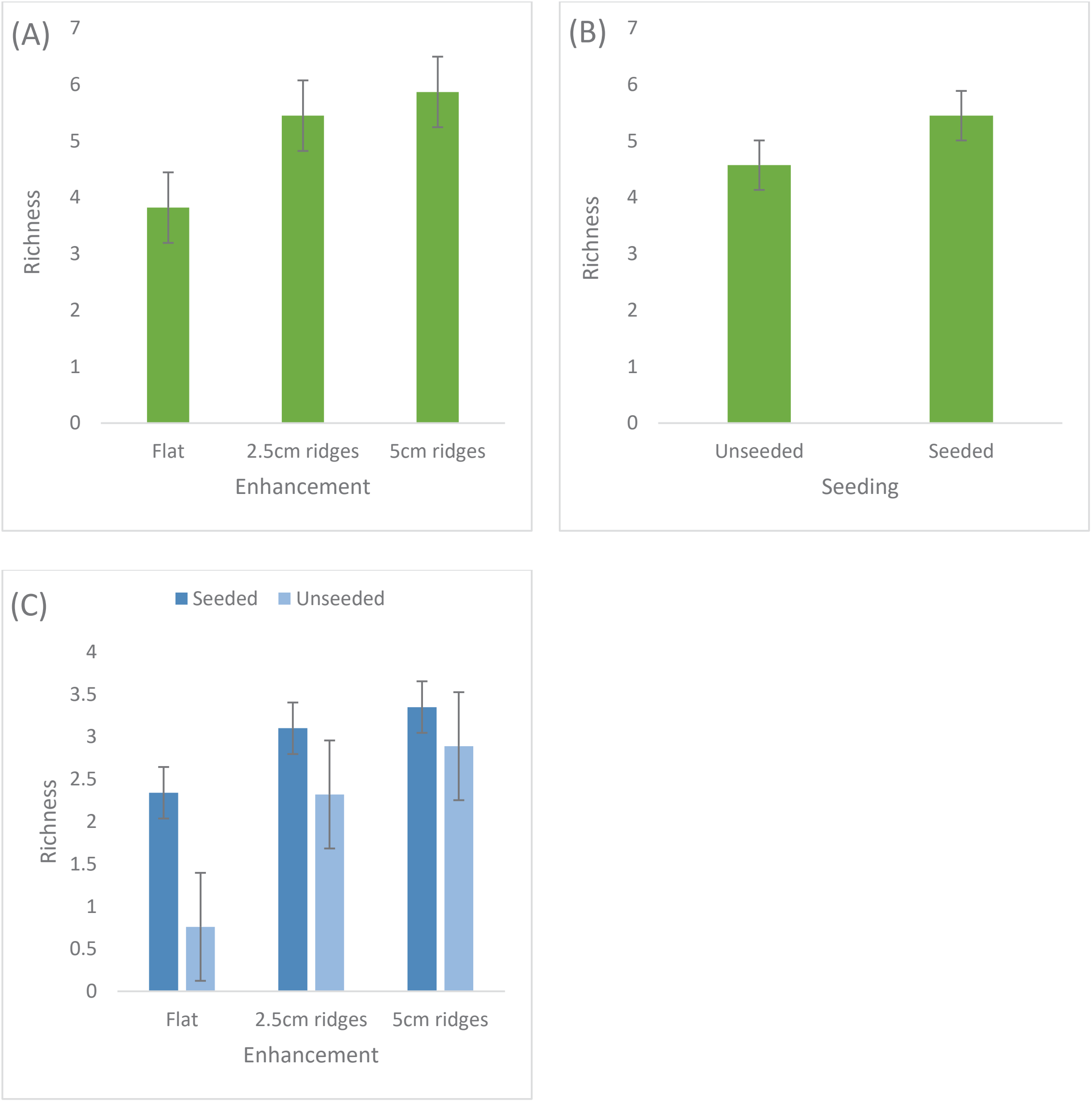

Figure 2.2: Effects of enhancement (A) and seeding (B) on the species richness of the fouling community as well as the interactive effects of enhancement and seeding on the mobile invertebrate community's richness (C). All plots denote means ± standard errors.

Additional complexity significantly affected the mean richness of the species found on the plates. The fouling communities (Fig. 2.2A and 2.2B) were positively influenced by complexity and seeding though the effects of these two factors were independent. For mobile invertebrate richness on the other hand, we found a marginally non-significant enhancement and seeding interaction P-value = 0.059; Appendix, Table 2. The significant enhancement by seeding interaction suggested that benefits of seeding for species richness declined for greater levels of plate complexity (Fig. 2.2C).

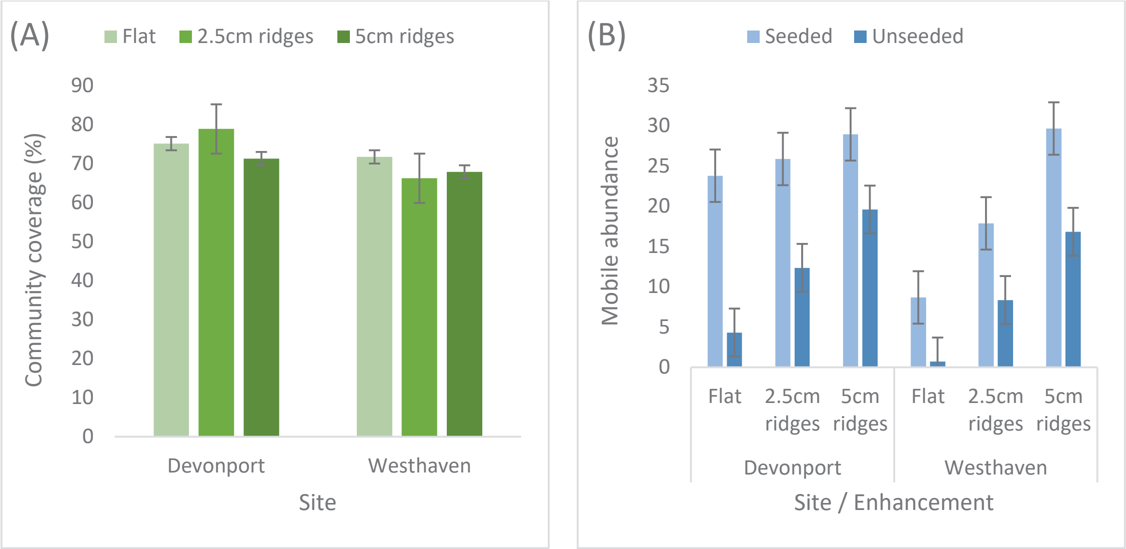

Figure 2.3: The interactive effects of enhancement and location on the fouling community coverage (A) as well as the interactive effects of enhancement and seeding on mobile invertebrate abundance (B). All plots denote means ± standard errors.

The complexity significantly affected the coverage of the fouling community as well as the abundance of mobile species, though there were some interactions with other factors. For the coverage of the fouling communities, the flat plates had greater coverage than the 5cm plates though the relative effects of the 2.5cm plate on coverage differed among locations, with greater coverage at Devonport but lower coverage at Westhaven (Fig. 2.3A).

For the mobile species abundance on the settlement plates, we found an interaction between the complexity, seeding, and location (Fig. 2.3B). Both enhancement and seeding increased the mean abundance of mobile species on a plate. However, at Westhaven the abundance increased fairly evenly across the plates, whereas at Devonport the abundance of mobile invertebrates was higher overall and the increase in abundance on seeded plates was greater on the flat plates than on the plates with enhancement.

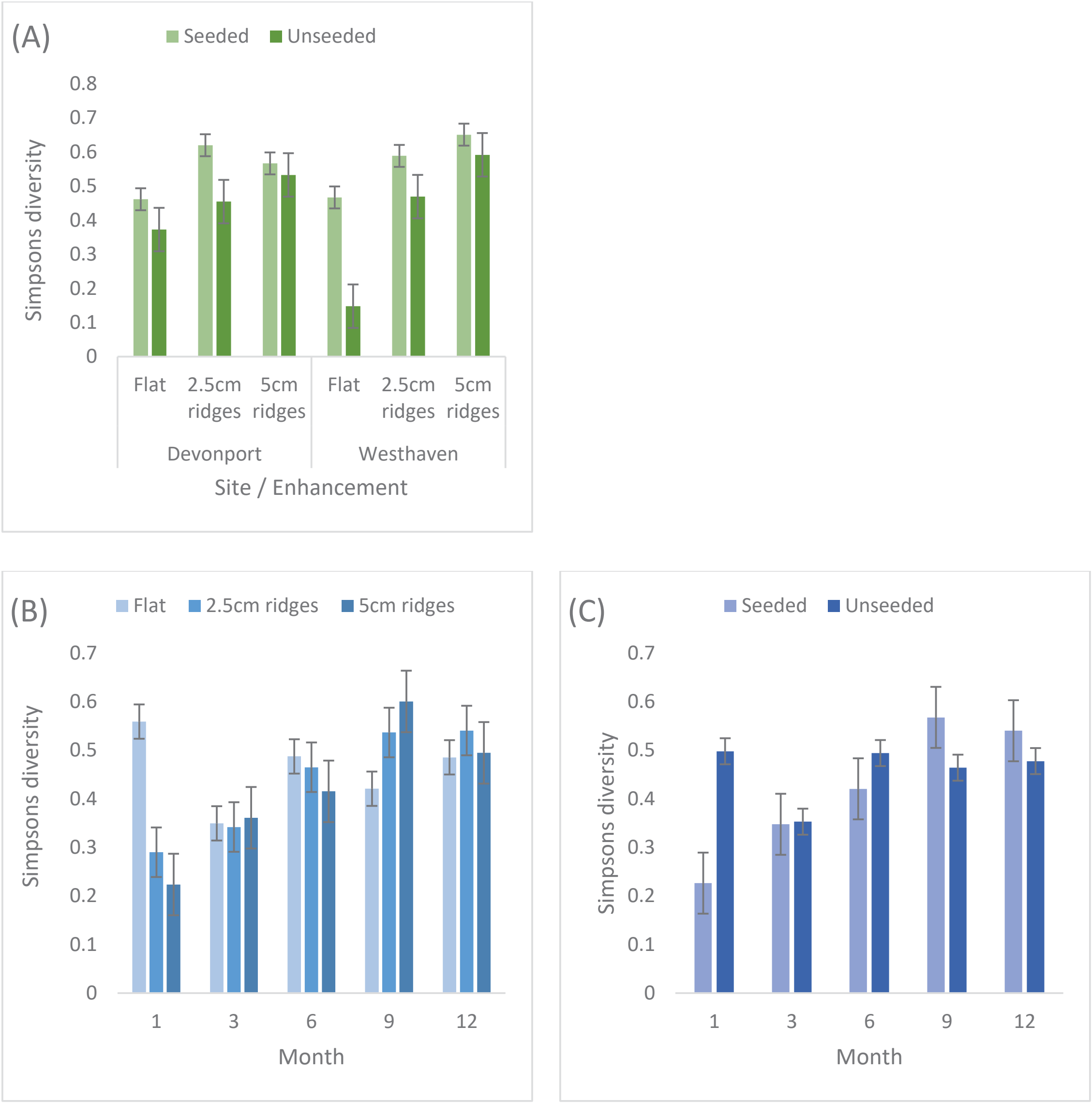

Figure 2.4: Effects of enhancement and seeding on the Simpsons diversity of the fouling community (A) as well as the separate interactive effects of month with enhancement (B) and seeding (C) on mobile invertebrates Simpsons diversity. All plots denote means ± standard errors.

The Simpsons diversity of the fouling community was influenced by the complexity of the plates and seeding, though the interaction differed between locations (Fig. 2.4A). Simpsons diversity increased with the addition of seeding and enhancement complexity at both locations, however the unseeded flat plates at Westhaven were significantly less diverse than all other treatment combinations.

For the Simpsons diversity of the mobile community, there was a significant interaction between enhancement and month as well as between seeding and month (Figs. 2.4B, 2.4C). However, while the Simpson diversity generally increased from month to month, the treatments effect showed no discernible pattern.

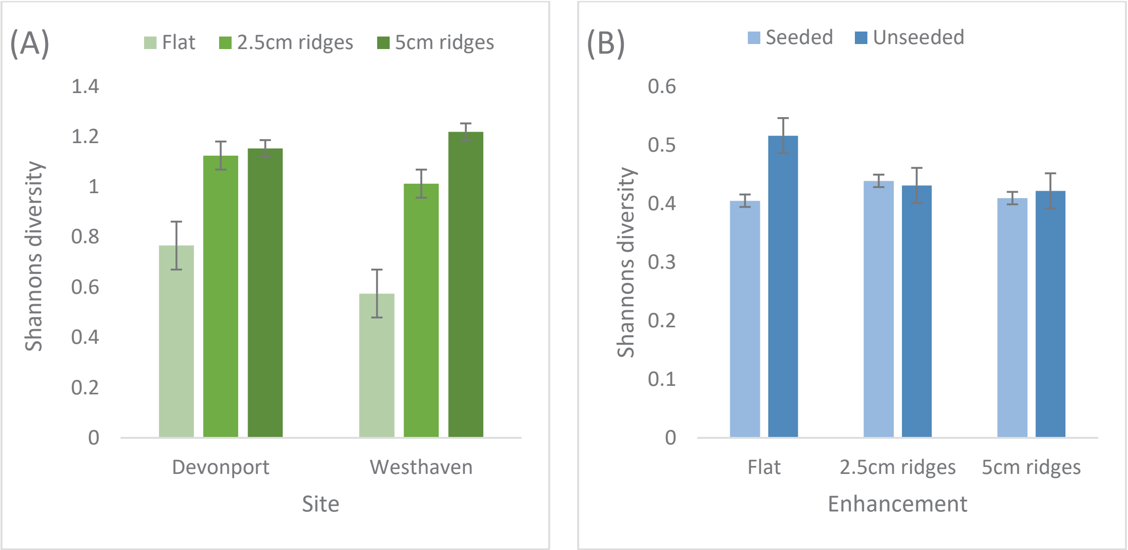

Figure 2.5: Effects of the interactive effects enhancement of and location on the fouling community's Shannon diversity (A) as well as the interactive effects of enhancement and seeding on mobile invertebrates Shannon diversity (B). All plots denote means ± standard errors.

For fouling communities, we found a positive effect of enhancement on Shannon's diversity, though the effect was strongest at Westhaven (Fig. 2.5A).

Alternatively, the Shannon's diversity of the mobile community was influenced by the interaction of seeding and enhancement, though the interaction appeared driven primarily by an unusually large Shannon's diversity on flat unseeded plates (Fig. 2.5B).

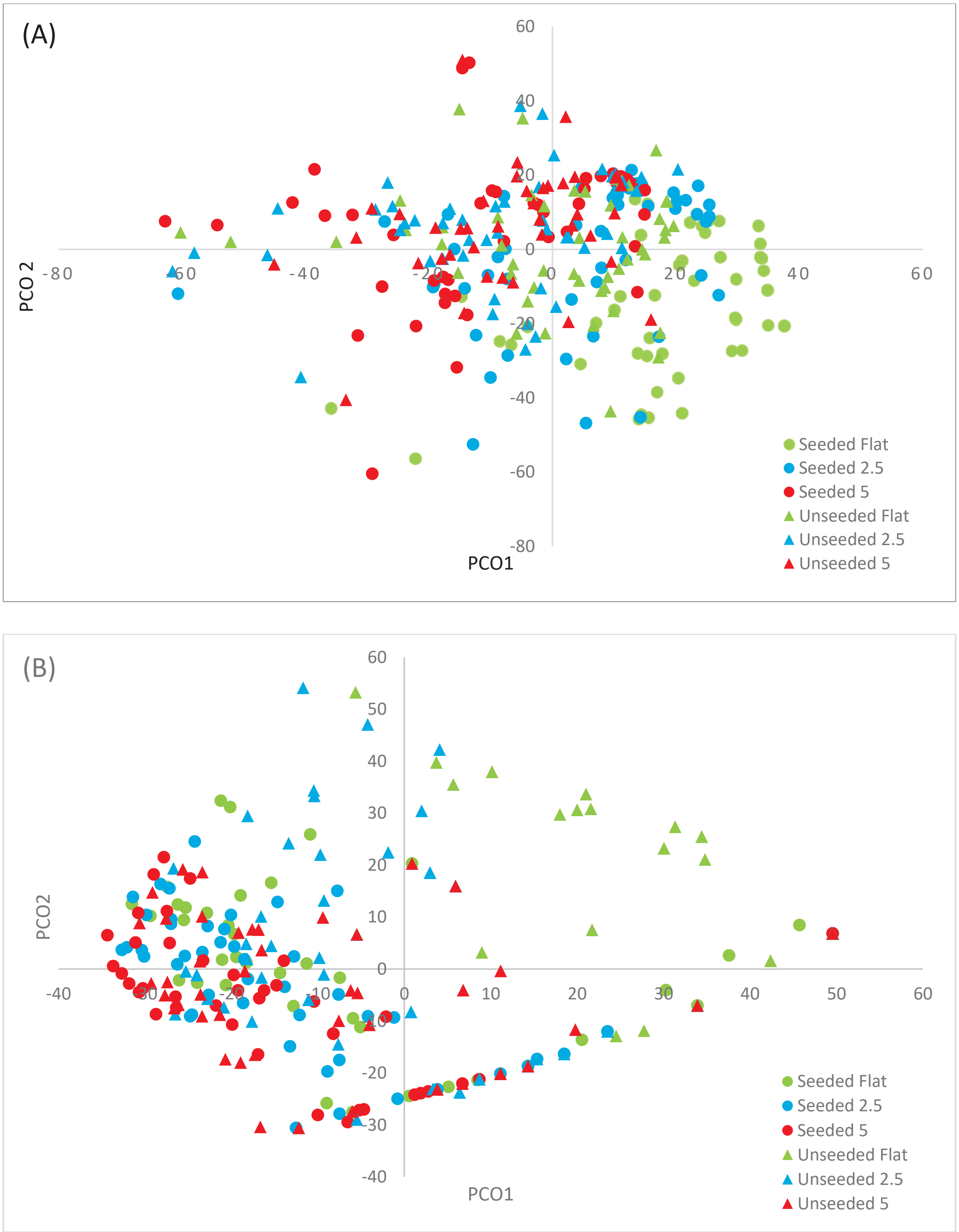

Figure 2.6: Bray Curtis dissimilarity of the Fouling community (A) and mobile invertebrate community (B). The first and second principal coordinates (PC1 and PC2) captured 26.1% and 23.1% of the dissimilarity respectively for the fouling community and 50.6% and 22% for the mobile community.

The fouling community composition of the settlement plates, expressed as Bray-Curtis dissimilarity, differed significantly among combinations of seeding and enhancement (Fig. 2.6A). Examining these differences in closer detail revealed that the interaction was driven by significant differences in community composition between the flat and ridged unseeded plates (Appendix. 1.22). Alternatively, for seeded plates, we only found a significant difference in community composition between the flat and 5cm ridged plates. Permutational analysis of multivariate dispersions (Anderson *et al.* 2005) confirmed that the multivariate dispersions of the data for each treatment combination were not significantly different ($F_{1,5}$, F=0.814 P (permutation) =0.691).

For the mobile communities, the interaction between enhancement and seeding also had a significant effect on community composition (Fig. 2.6B). In their case, the community composition of seeded plates was not significant different among enhancement types. However, in the absence of seeding the mobile communities were significant distinct between the flat and ridge plates (Appendix. 1.23). This result however should be interpreted with the caveat in mind that the multivariate dispersions were not constant among treatment combinations ($F_{1,5}$, F=2.736 P=0.033*).

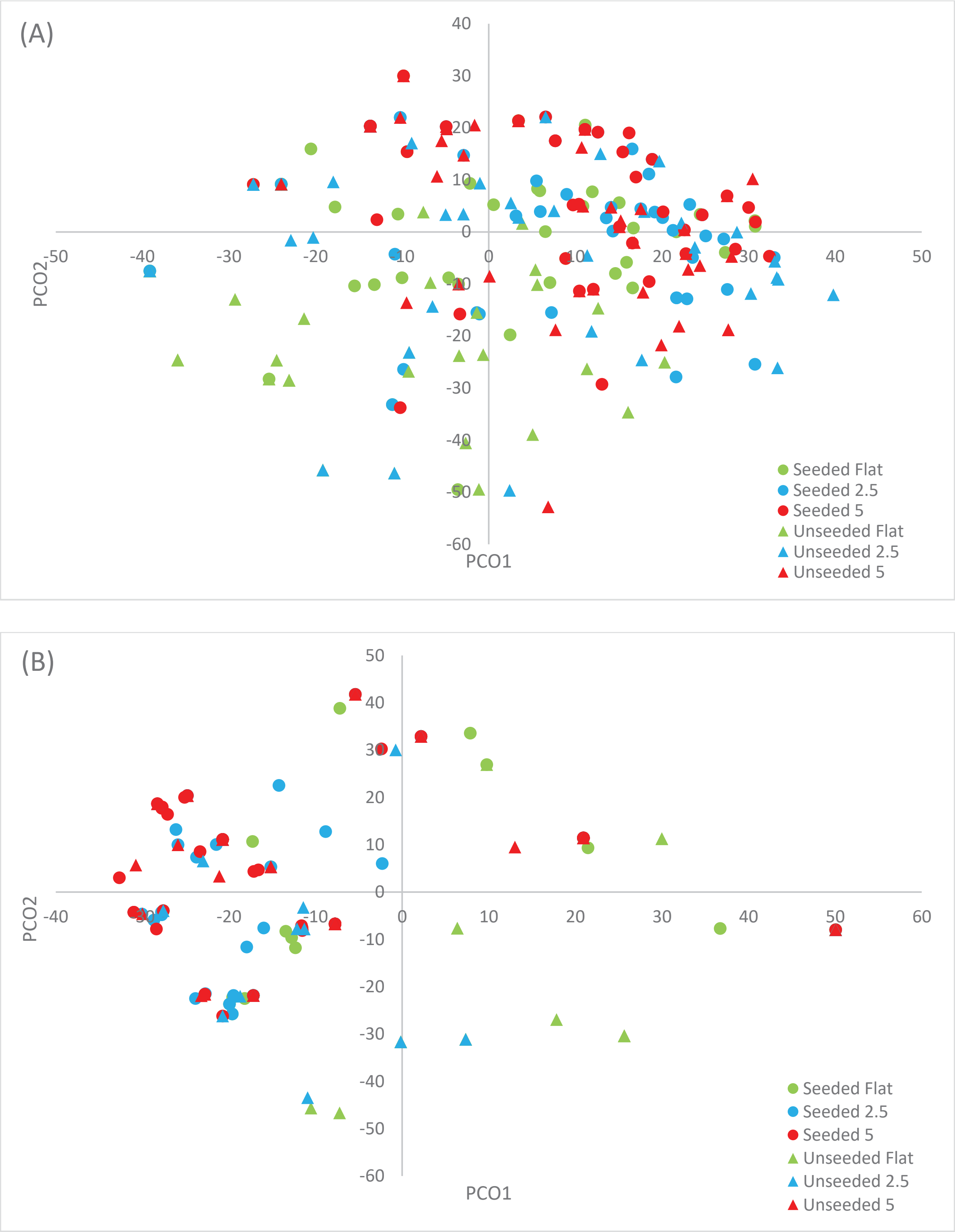

Figure 2.7: Jaccards similarity coefficient of the Fouling community (A) and mobile invertebrate community (B). The first and second principal coordinates (PC1 and PC2) captured 24.6% and 23.1% of the dissimilarity respectively for the fouling community and 39% and 21.3% for the mobile community.

For both the fouling community (Fig. 2.7A) and the mobile community (Fig. 2.7B) the community composition, expressed a Jaccard's dissimilarity, of plates was significantly affected by the interaction of seeding and enhancement. In both communities, the composition was only different between unseeded flat and ridge plates (Appendix. 1.24 & 1.25). Permutational analysis of multivariate dispersions of each community confirmed that the multivariate dispersions of the data for each treatment combination we not significantly different, $F_{1,5}$, F=0.748 P=0.718(A) and $F_{1,5}$, F=0.1.21 P=0.387(B).

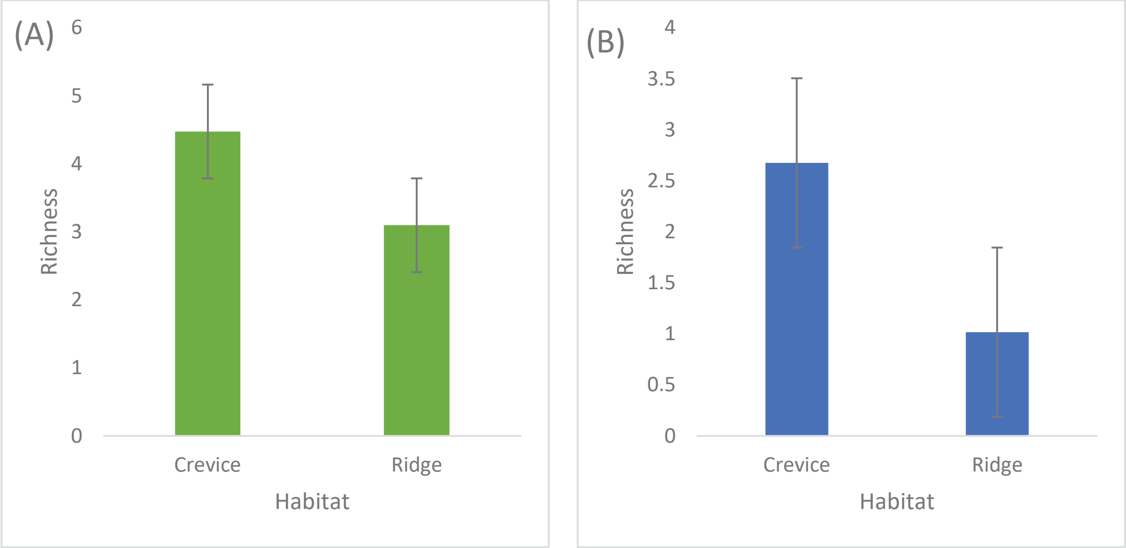

Figure 2.8: Main effects of the habitat utilisation on the fouling community's richness (A) as well as the effect of habitat utilisation on mobile invertebrate's richness (B). All plots denote means ± standard errors.

The differences between crevice and ridge habitats of the ridged settlement plate significantly affected the biodiversity of the fouling and mobile communities. Both communities saw increases in mean species richness (Figure 2.8) and Shannon's diversity (Figure 2.10), as well as increases in abundance (Figure 2.9) and Simpsons diversity (Figure 2.11) for only the mobile and fouling communities respectively.

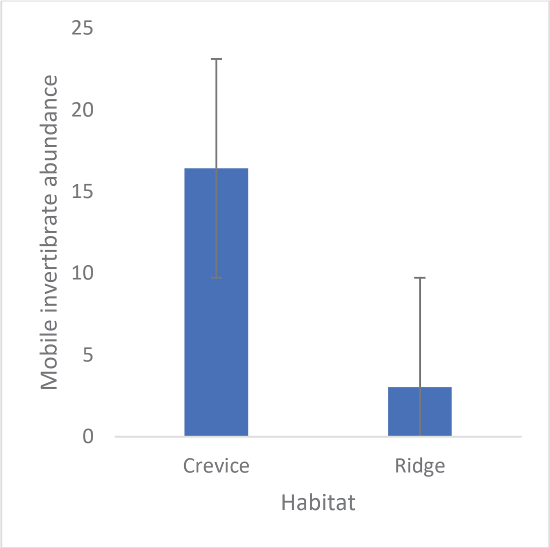

Figure 2.9: Main effects of the habitat utilisation on the mobile invertebrate's abundance. All plots denote means ± standard errors.

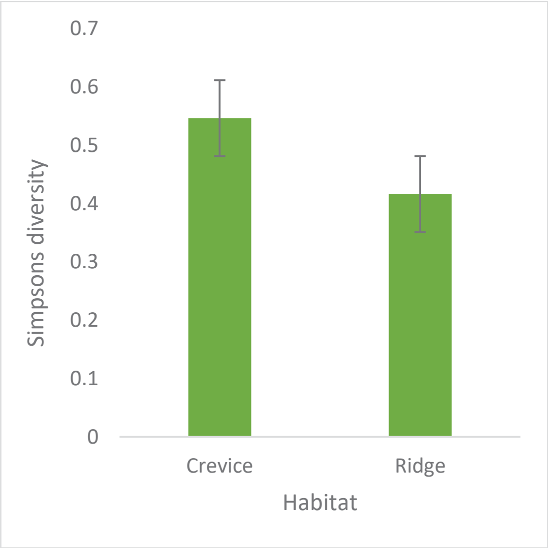

Figure 2.10: Main effects of the habitat utilisation on the fouling community's Simpson diversity. All plots denote means ± standard errors.

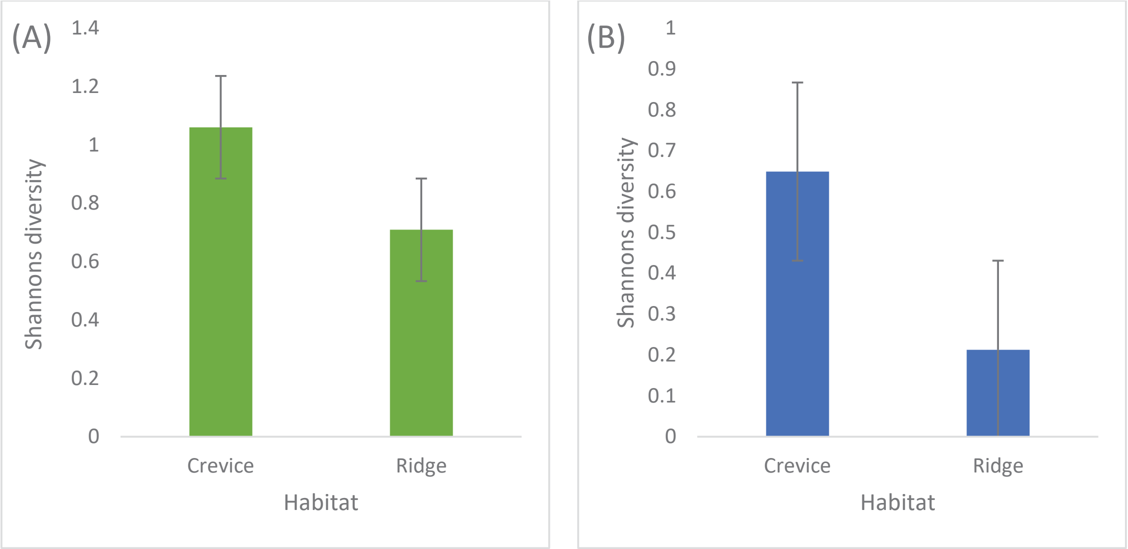

Figure 2.11: Main effects of the habitat utilisation on the fouling community's Shannons diversity (A) as well as the effect of habitat utilisation on mobile invertebrates Shannons diversity (B). All plots denote means ± standard errors.

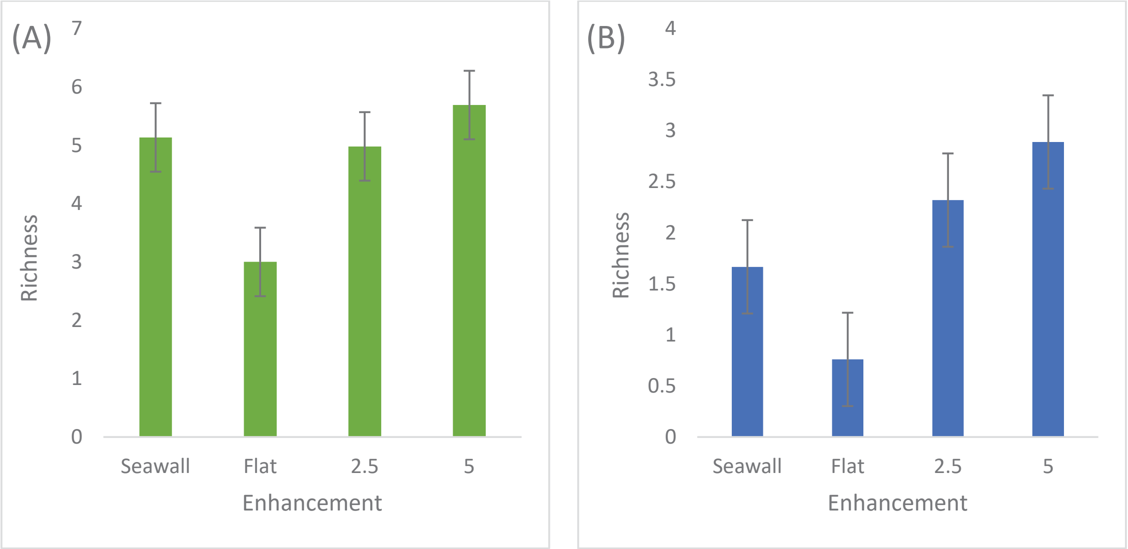

Figure 2.12: Main effects of the enhancement of the fouling community's richness with the seawall control (A) as well as the effect of enhancement on mobile invertebrate richness with the seawall control (B). All plots denote means ± standard errors.

Comparing the eco-engineered plates and areas of traditional seawall of the same area we found the mean richness (Fig. 2.12) of the mobile and fouling species and the abundance (Fig 2.13) of mobile species on the traditional seawall was greater that the flat plates but similar to that of the 2.5cm ridged plates. For the Simpson's (Fig. 2.14) and Shannon's (Fig. 2.15) diversity of fouling communities, the similar pattern occurred where the traditional seawall had similar biodiversity to the ridged plates, though the effect differed among locations. For the mobile invertebrate communities on the other hand, mobile invertebrate communities had greater Simpson's diversity on the flat plates than the ridged plates and seawall whereas for Shannon's diversity the opposite pattern was observed.

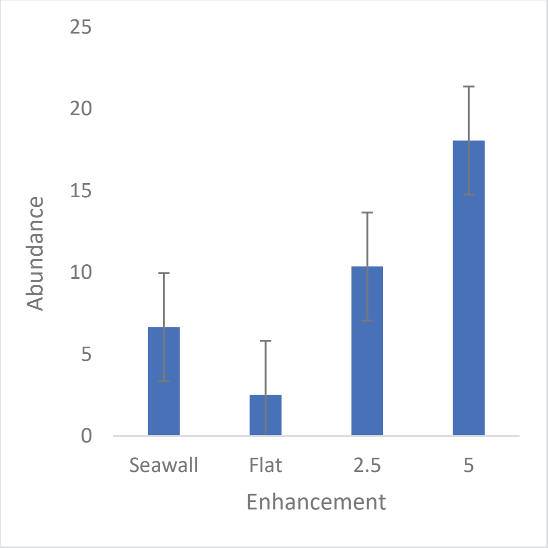

Figure 2.13: Main effects of the enhancement on mobile invertebrates abundance with the seawall control (B). All plots denote means ± standard errors.

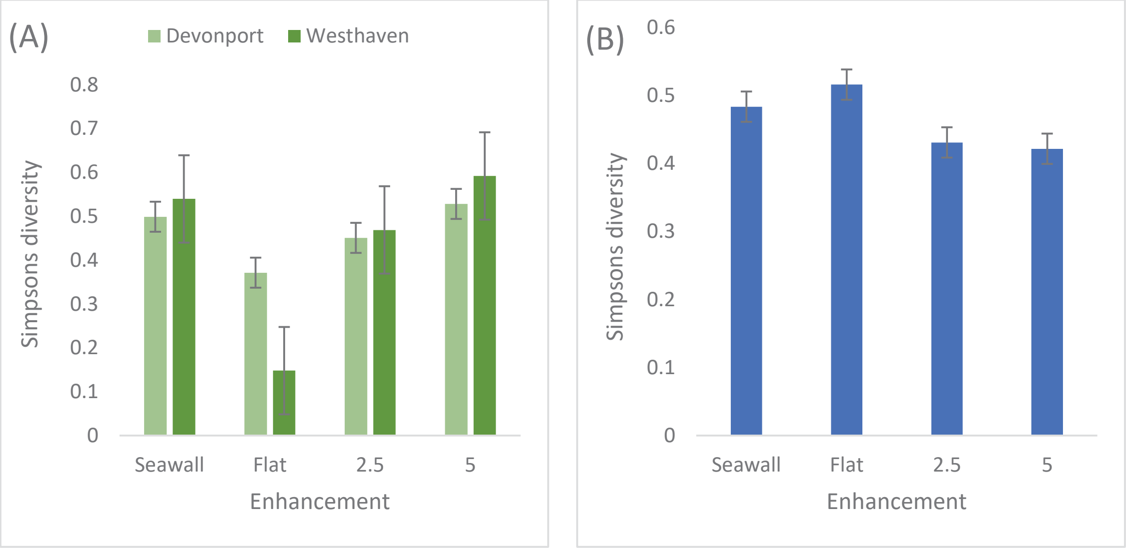

Figure 2.14: Main effects of the enhancement and location the fouling communities Simpsons diversity with the seawall control (A) as well as the effect of enhancement on mobile

invertebrates Simpsons diversity with the seawall control (B). All plots denote means ± standard errors.

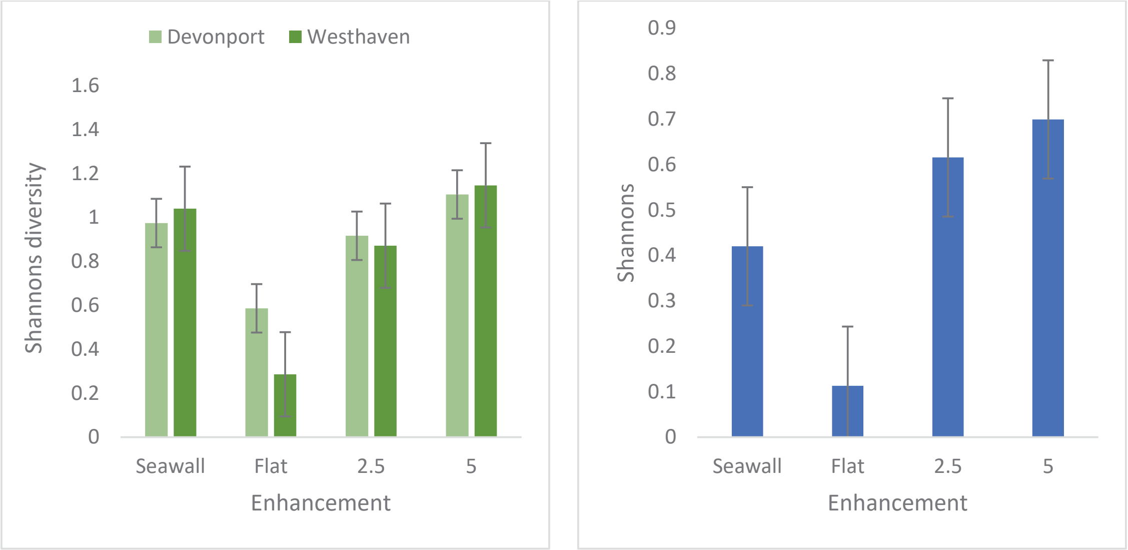

Figure 2.15: Main effects of the enhancement and location the fouling communities Shannon diversity with the seawall control (A) as well as the effect of enhancement on mobile invertebrates Shannons diversity with the seawall control (B). All plots denote means ± standard errors.

Figure 2.16: Bray-Curtis dissimilarity of the Fouling community (A) and mobile invertebrate community (B). The first and second principal coordinates (PC1 and PC2) captured 41.8% and 21.7% of the dissimilarity respectively for the fouling community and 54.8% and 24.1% for the mobile community.

For fouling communities (Fig. 2.16A), the Bray-Curtis test found significant difference between the plates and seawall communities, interacting with site. The community composition of the plates was significantly different to the traditional seawall in every case, except for the 2.5cm plates at Devonport (Appendix. 1.26). This result however should be interpreted with the caveat in mind that the multivariate dispersions were not constant among treatment combinations ($F_{1,3}$, F=10.359 P=0.000*).

For mobile communities (Fig. 2.16B), significant difference was found between the seawall and enhancements interacting location and month. As the most developed community, the 12-month statistics was used as the example (Appendix. 1.27). While the seawalls mobile community was significantly distinct between the plates at Westhaven, at Devonport, the ridged plates did not have a significantly different community. Permutational analysis of multivariate dispersions confirmed that the multivariate dispersions of the data for each treatment combination we not significantly different ($F_{1,3}$, F=0.884 P=0.494).

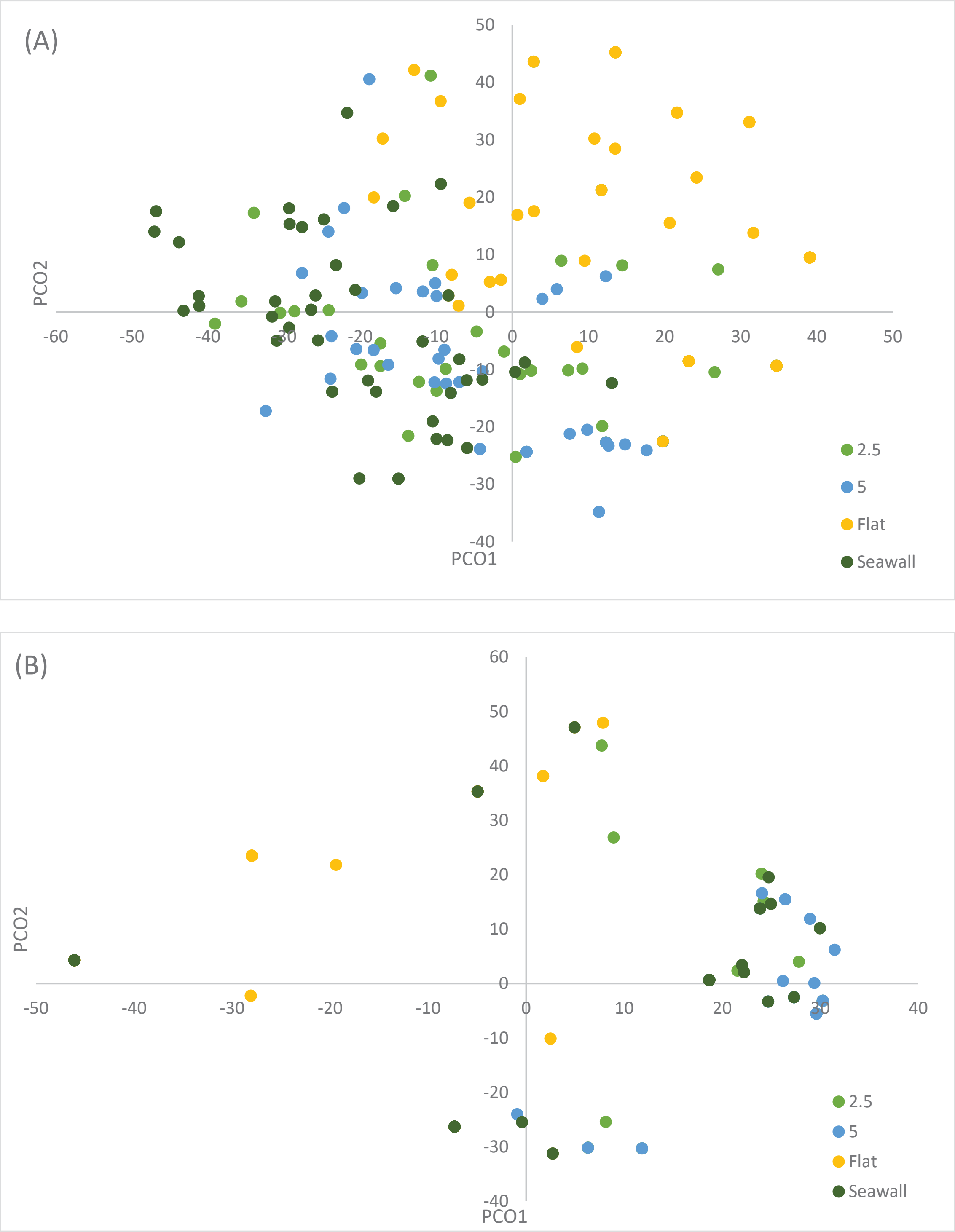

Figure 2.17: Jaccards similarity coefficient of the Fouling community (A) and mobile invertebrate community (B). The first and second principal coordinates (PC1 and PC2) captured 27.8% and 21.6% of the dissimilarity respectively for the fouling community and 46.3% and 23% for the mobile community.

For both the fouling community (Fig. 2.17A) and the mobile community (Fig. 2.17B) the community composition, expressed a Jaccard's dissimilarity, between the seawall and plates was significantly different, interacting with location and month. As the most developed community, the 12-month statistics was used as the example (Appendix. 1.28, 1.29). While the seawalls communities were significantly distinct between the plates at Westhaven, at Devonport, the ridged plates did not have significantly different communities. (Figs. 16A, 16B). Permutational analysis of multivariate dispersions of each community confirmed that the multivariate dispersions of the data for each treatment combination we not significantly different, $F_{1,5}$, $F_{1,3}$, F=0.884 P=0.494(A) and $F_{1,3}$, F=1.027 P=0.44 (B).

2.5 Discussion

Results support the paradigm that habitat heterogeneity increases biodiversity. The complexity introduced by ridges and the seeding of mussels both significantly increased the biodiversity of the fouling and mobile invertebrate communities. Often, enhancement and seeding complexity complimentary positively influenced biodiversity. Importantly, but perhaps not surprisingly, seeding often had a greatest effect on biodiversity on flat plates where the seeded mussels provided the only habitat complexity on the plate. While fouling community coverage was unaffected by the additional complexity, the fouling communities on the more complex treatments were distinctly more diverse. These results suggest that the artificial seeding of seawalls or incorporating physical complexity into their design would greatly enhance the local biodiversity over flat, uniform designs.

As the mussels were attached on their side to avoid gluing their valves shut, the mussels created a small crevice like habitat between their shells and the plate (Photo 2.2). This gap provides excellent protection for mobile invertebrates to inhabit during low tides, therefore having the greatest benefited to the mobile community. As a result, the plate seeding benefited every measure of diversity within the mobile communities, which was not replicated by the fouling community. However, our artificial seeding method resulted in the mussels being placed on their side, creating the maximum area of crevice environment. This positioning does not occur within natural mussel populations, therefore, the influence that the mussels had on the seawalls diversity was maximised by this experiment.

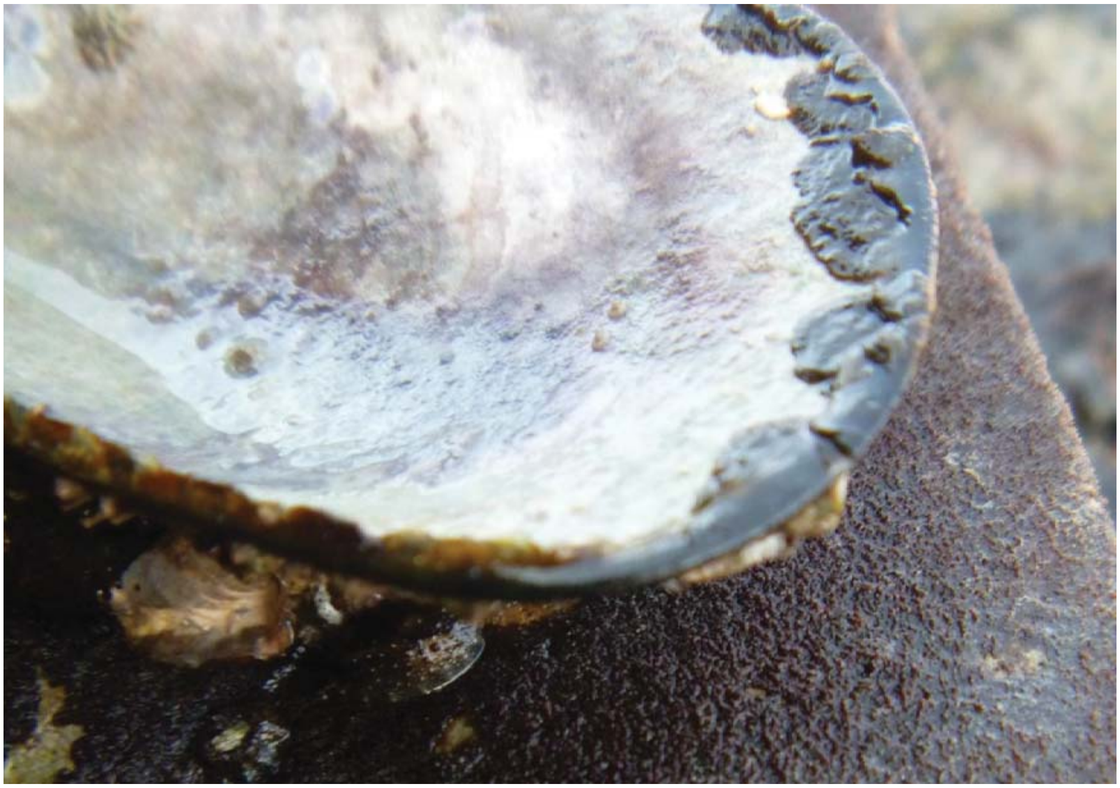

Photo 2.2: Leather slug (*Onchidella nigricans*) utilising the habitat under the rim of a dead seeded mussel (*Perna canaliculus*) at Westhaven. Photographer: Connor M^{c}Kenzie

Generally, the fouling and mobile invertebrate communities of Westhaven were less biodiverse than Devonport. However, despite these effects of location, typically the benefits of complexity (seeding and enhancement) manifested at both site with their interaction driven by different magnitudes. This supports that additional seawall complexity results in more diverse and distinct communities, regardless of location.

The biodiversity of crevice habitats was consistently greater than ridged habitats on complex plates. The only exception was with the coverage of the fouling communities and the Simpsons diversity of the mobile communities. Interestingly, though seeding typically enhanced biodiversity, the effects of seeding did not generally interact with habitat utilisation (tables 2.1, 2.2), suggesting that the seeding enhanced diversity regardless of whether they were in a crevice or an exposed ridge. This is further supported by the Bray-Curtis, and Jaccards diversity tests, as the communities were significantly distinct between the flat and complex plates when unseeded, but the introduction of seeding raised the diversity across the plates to a point where the communities were not significantly different. An exception is the Bray-Curtis between the flat and 5cm seeded plates, which are significantly diversity. This shows evidence that the seeded complex plates still obtained a significantly different community than the communities upon flat seeded plate.

The sections of traditional seawall, which were intended to be a control, often had more biodiversity than the flat eco-engineered plates. The most common seawall material in Auckland and that was used at our experimental sites, is not ubiquitous flat concrete surface, but rather a cemented pavement of volcanic rocks. Therefore, the traditional seawalls used as controls in this experiment has significant fine-scale complexity. Furthermore, because of the presence of small pits and holes in the volcanic rock, the wire brush was unable to remove all the prior community, leaving a reservoir of spores or fragments from which the community could re-establish. Nevertheless, although the eco-engineered plates were devoid of any organisms at the commencement of the experiment the traditional seawall typically only achieved a biodiversity similar to that of the 2.5cm ridged plates. Both the Bray-Curtis and Jaccards dissimilarly measures, showed that the communities on the ridged plates were distinct from the communities inhabiting seawalls at Westhaven, and the 5cm ridged plates at Devonport. Overall, these results confirm that the level of complexity on a seawall, increases diversity of the community.

Some of the differences between the patterns of biodiversity found at Westhaven and Devonport could have been due to the poor survival of seeded mussels at Westhaven. The seawall at Westhaven is the breakwater for New Zealand's largest marina, as is likely to have high concentrations of pollutants from marine traffic, such as oil spills and anti-foul leaching which are known to have a detrimental effect on the diversity of local fouling communities (Piola & Johnston, 2008). Despite the existence of location effects for some measures, the location effects were not significant for many biodiversity metrics. Therefore, the benefits of spatial heterogeneity occur across locations, including seawall communities exposed to greater likelihood of pollution from human activities.

At the conclusion of the experiment, apart from *Crassostrea gigas* which has naturalised on Auckland's seawalls, only one NIS was identified on our settlement plates. The bryozoan *Watersipora* sp. was found on a ridged, seeded plate at Devonport. At least 102 NIS have been identified within the Waitematā Harbour (Chapter 4. Results), and in-depth examination of fouling communities in the Ports of Auckland found 12 NIS. The seawall and pilings that make up the Ports of Auckland are a uniform concrete seawall and the area is extremely polluted (Inglis *et al.* 2006). However, these species were not strongly established on the more complex volcanic rock seawalls used in this experiment or the eco-engineered plates. Therefore, the results are a positive indication that Auckland harbours seawalls are an improvement on uniform concrete seawalls, however, eco-engineered seawalls are still the best option for retaining the greatest biodiversity.

The overwhelming consensus of our results supported that additionally spatial heterogeneity increased biodiversity on intertidal communities. Current seawall communities are highly degraded as a response to uninspired uniform designs and these less diverse communities are therefore at a greater risk of invasion by NIS. This study confirmed that these detrimental effects can be mitigated, through the implementation of intelligent complex seawall designs constructed from eco-friendly concrete blends. Furthermore, the artificial seeding of native bivalves can independently increase the biodiversity of the seawall community and could be utilised on existing modifications. The combined implementations of these practices in metropolitan harbours can therefore significantly reduce the circum-global degradation of urban foreshores.

Chapter 3: The implications of climate change driven increases in rainfall on the communities inhabiting ecoengineered substrates in the mid-intertidal zone.

3.1 Abstract

The global climate is changing at an alarming rate and calibrating the effects of climate change on biodiversity is one of the most pressing concerns in contemporary ecology. Climate change will impact communities in multiple ways by affecting temperature, precipitation, wave energy and ocean chemistry. While some of the climate change drive changes in environmental conditions have received a lot of attention (e.g. temperature and ocean chemistry), we know very little about the likely impacts of other factors. Here, we examined the potential consequences of climate change driven increases in precipitation on seawall biodiversity. Moreover, we examined the interaction between precipitation and ecoengineered substrates of varying complexity. The influence of habitat complexity on biodiversity was reaffirmed; however, we also found the potential for a positive relationship between rainfall and the fouling community biodiversity, possibly as a response to lower desiccation stress. The proposed relationship would result in a vertical shift in the distribution of species from the low intertidal to mid intertidal. Therefore, an increase in rainfall would raise competition of seawalls resources, forcing outcompeted species to shift their horizontal distribution.

3.2 Introduction

Anthropological influences are increasing the rate of global climate change far beyond the rate of natural climatic oscillations. As a result, climate change is having causing unprecedentedly rapid environmental change and threatening to destroy whole ecosystems which cannot adapt to the new environmental regimes. The full extent of anthropologic climate change effects is controversial, but the scientific consensus acknowledges it as one of the greatest issues faced by the planet, as well as humanity, and requires considerable monitoring and mitigation efforts.

Anthropogenic climate change is driven by the enhanced output and reduced recycling of greenhouse gases within the atmosphere resulting from human activities and changes in land use. Common anthropogenic greenhouse gases include carbon dioxide (CO_2), Methane (CH_4), Nitrous oxide (N_2O), and fluorocarbons. Greenhouse gases absorb and emit radiation, therefore solar energy that penetrates the earth's atmosphere and is reflected off the earth's surface can be intercepted by greenhouse gasses, retaining the energy within the atmosphere rather than allowing it to pass back into space (Houghton *et al.,* 1992). While these gases are imperative for the retention of heat, increasing concentrations of these gasses from anthropogenic sources has caused global warming at an unprecedented rate (Stocker, 2014).

Since the beginning of the industrial revolution (1750 a.d.), atmospheric concentrations of carbon dioxide have increased by 40% (Tans & Keeling, 2017). Increases in CO_2 are primarily the result of a combustion of fossil fuels, such as coal, oil, and natural gas. In 2013, it was estimated that anthropogenic sources emitted 34.4 billion tons of CO_2 into the atmosphere (CAIT, 2017). As temperatures rise from the greenhouse effect, our polar ice caps are being reduced which creates a negative feedback loop of heating as CO_2 stored within the ice is released, and as the ice caps reflect solar radiation far more efficiently than water (Perovich *et al.,* 2007). This rising CO_2 is negatively complimented by mass deforestation for timber and land use. global deforestation has resulted in a fraction of the original forest coverage remaining in developed countries and the loss of millions of hectares of remaining tropical rainforests every year (Achard *et al*., 2002). Furthermore, the abundant peat located within forest are an abundant sink for CO_2 and slash and burn techniques used to clear large area, result in the release of this $CO_{2,}$ along with the forest biomass (Indonesia, 2008). As a natural CO_2 recycling source, converting it into oxygen (O_2), it's ironic that the global deforestation is attributing to some of the greatest CO_2 emissions, by some estimates accounting for up to a third of total anthropogenic emissions (Fearnside & Laurance, 2004). While some forests are later reseeded, much deforestation results in a conversion of land use into agricultural pasture (Noble *et al.,* 2000) The overwhelming contribution of CO_2 humans are emitting is accumulating an enormous carbon debt, which we now 'owe' the planet.

The mass rearing of livestock is another concerning contributor to anthropogenic climate change. Outputs produced through rearing livestock include the greenhouse gas Nitrous oxide (N_2O) as a by-product of fertilisers and methane gas (CH_4) being biologically produced by the livestock (Steinfeld *et al.,* 2006). While CO_2 is by far the most abundant greenhouse gas emitted by humans, other greenhouse gases are far more effective at retaining radiation in the atmosphere, making them significant concerns for climate change, despite their lower anthropogenic emissions (Forster *et al.,* 2007). For example, over a 20-year time frame the direct radiative effect of a mass of methane is about 72 times stronger than the same mass of carbon dioxide (Forster *et al.,* 2007). With 104 million tons of CH_4 being produced by livestock annually (Steinfeld *et al.,* 2006), over a 20-year timeframe the contribution to the greenhouse effect of methane is nearly equivalent as emitting an additional 7.5 billion tons of CO_2. As the human population increases, emission levels will only continue to rise, more land will be cleared, and livestock populations will increase unless radical action is taken to alternate to cleaner energy sources, permanent reforestation and efficient farming practices.

Anthropogenic Climate change generally results in an increase in global air and sea surface temperatures, changes in precipitation patterns, more frequent droughts and heatwaves, increasingly power storm systems, ocean acidification, and reducing glaciers and polar ice caps to raise sea levels (Stocker, 2014). As a sink for atmospheric CO_2, the ocean absorbs around 25% of the CO_2 emitted from anthropogenic sources (Canadell *et al.,* 2007). However, this CO_2 reacts within the water, being converted to carbonic acid which dissociates into bicarbonate to release an H^+ ion (Hoegh-Guldberg *et al.,* 2007). The collective accumulation of this reaction is gradually increasing the oceans acidity in correlation to atmospheric CO_2. These H^+ ions additionally react with carbonate to reform bicarbonate, ultimately reducing carbonate quantities in the ocean (Raven *et al.,* 2005). As carbonate is utilised in the process of calcification by shellfish and corals, the reduction of carbonate is reducing the rate at which these organisms can grow (Raven *et al.,* 2005). Despite coral reefs being on of the most diverse environments in the marine system, corals themselves are very slow growing organisms. Therefore, the rapid drop in carbonate caused by acidification is resulting in the death of large areas of coral reef which are unable adapt to the sudden climatic shifts (Raven *et al.,* 2005). On land and sea, sensitive species, such as corals, are at immediate risk of local extinction in response to the effects of anthropogenic climate change.

As one of the most well studied natural environments, the rocky intertidal zone is speculated to be the perfect natural environment to monitor the effects of subtle climate changes that are occurring in both the terrestrial and aquatic environments (Helmuth *et al.,* 2006). The gradient of habitat complexity across the intertidal zone makes it a highly diverse and competitive environment, which has been utilised by researchers to examine interactions between ecological patterns, biotic interactions, and abiotic stresses in nature (Bertness *et al.,* 1999; Southward, 1958). At its upper limits, intertidal organisms are greatly affected by terrestrial climatic conditions and must endure desiccation and thermal stresses, while at the lower limits the community is more affected by oceanic conditions such as sea surface temperature, dissolved oxygen, upwelling regimes, and oceanic pH (Davenport & Davenport, 2005). Stressors resulting from climatic changes, result in both community shifts vertically along the intertidal height and horizontal shifts, with climate change potentially contracting their geographic distribution or conversely increase their distribution as environmental conditions at previously unoccupied sites become physiologically tolerable for the first time (Helmuth *et al.,* 2006). The observable reactions of subtle environmental changes to the intertidal community, indicates that the intertidal zone is an excellent natural environment for the monitoring of climate change effects.

The intertidal zone was one of the earliest study systems to be used as a model system to examine global warming, when distributions of intertidal barnacles over forty years was linked to be a result of ‘climatic deterioration’ (Southward, 1967). Since then, the intertidal zone has been a model to help understand a range of questions related to climate change including how distributions are affected by air and sea surface temperatures (Helmuth *et al.,* 2002; Somero, 2002), how close populations can survive to the limits of their physiological tolerances (Southward, 1958), how climate change indirectly affects community composition through modifications of biotic interactions (Sanford, 1999; Burnaford, 2004), and how rapidly populations can evolve in reaction to climate change (Clark, 2003). With global warming a foremost concern of climate change, many publications focus on the effect of thermal variation; however, the intertidal zone has the same potential to be utilised to test the effects of more subtle or underappreciated climate change effects.

This study utilises the intertidal zone to examine how changes in precipitation patterns affect the diversity of intertidal seawall communities. Moreover, we examine how green-engineering technologies could be used as a climate change mitigation strategy. This way we could see how a change in precipitation upon the seawall could affect the development of the seawall community and how complex eco-engineered environments may enhance or dampen climate change effects on these communities. A report to New Zealand's Ministry for the Environment in 2008 predicted that Auckland's summer rainfall could increase by up to 20% by 2090, compared to the rainfall recorded in the summer of 1990 by the Māngere rainfall station. Using this prediction, we simulated a 20% increase in rainfall over the summer period to examine the effects of climate change on community composition under conditions of enhanced precipitation.

3.3 Methodologies

3.3.1 Rainfall data

The rainfall data for Māngere rainfall station in 1990 was downloaded and sorted into the appropriate seasons. The total rainfall of the summer of 1990 was summed and was divided by the number of days in summer to produce the average daily rainfall and used as our baseline to calculate what a 20% increase in rainfall would be in. We then calculated the volume of water required to result in a 20% increase in daily rainfall for a 25cm × 25cm plate using a rainfall calculator (http://www.calctool.org/ CALC/other/default/rainfall). We then multiplied this value by three (53.75mL) and applied this volume of water to the plates every three days using a garden spray bottle that delivered the water as a fine mist directly to the plate. The water was distributed using the modified spray bottle (Photo. 3.1), spritzing 20cm from the highest surface of each plate and shifting the spritzer around the plate to ensure an approximately even distribution, while keeping the cone of water spray within the edges of the plate.

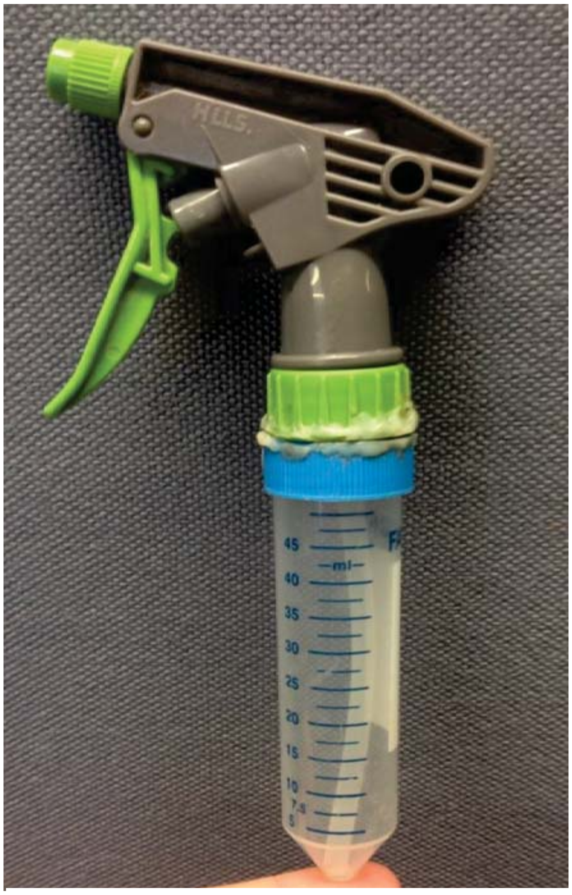

Photo 3.1: Modified spray bottle used to apply additional ‘rainfall’.
Photographer: Connor M^{c}Kenzie

3.3.2 Seawall preparation

The experimental design used in Chapter two was replicated for this experiment, with some minor modifications. The Devonport seawall was selected as the location. The tidal height for the deployment of the plates was raised by approximately forty centimetres relative to chapter twos experiment, placing the plates 1.2m above chart datum. At this tidal height, the plates were exposed at every low tide.

Thirty 25x25cm plots, no less than one meter apart along the seawall were scoured and wire brushed to remove existing fouling organisms. Two 8mm wide, 70cm deep holes were drilled in diagonally opposite corners of each scrapped plot on the seawall for attachment of the settlement plates. A randomised template was created to determine which treatment will be assigned to each plot. The plates were attached with the ridges in a horizontal orientation, to retain water, using two stainless steel dynabolts (8mm x 80mm) with an 8mm nylon and stainless-steel split washer between the plate and the dynabolt hex nut. Two-part Splash Zone Compound, Underwater Epoxy Putty, A-788 was used as an additional adhesive behind and around some edges of the plates. The plates were deployed into the field on the first day of summer (01/12/16) with the artificial 'rainfalls' occurring every 3rd day afterwards.

3.3.3 Monitoring

The plates were monitored monthly over the summer. At the end of each month the plates had two replicate photos, 50cm from the front of the surface of each plate were taken to ensure that we obtained an in-focus picture coving the entire treatment area, which would be used during later photo analysis. Several photos of each plate would then be taken at various perspectives with focus on highly diverse areas, mobile invertebrates and uncommon / rare species. The photo analysis was performed using Coral Point Count with Excel extensions program which overlaid the area of the plates with random points. Upon each plate, 100 points were used to identify species presence and estimate the coverage of the fouling community. On the complex plates, 50 random points each were used from the areas on the ridge and in the crevices.

All mobile invertebrates on a plate were identified, counted, and recorded. Any new species found that were not immediately identifiable were assigned an operational taxonomic unit until their photos could referred to experts for identification. Three areas of each plate were ignored for physical and photo identification: on the bolt ends, hexnuts, and washers; within the two remaining empty bolt holes; and the first 2cm of the base of each plate (i.e. the base of the flat plates).

3.3.4 Seawall deconstruction

After the third and final round of sampling, the seawall was returned to its original condition. Each settlement plate was removed from the seawall by removing the nuts and dynabolts. The plates were then carefully removed from the seawall and were scoured to remove the fouling that had accumulated upon the back of each plate. Each plate was then placed into a labelled plastic bags and carefully packed into plastic storage containers for transport back to the laboratory where they were frozen before sorting.

3.3.5 Statistical Analysis

Our analysis considered enhancement as a fixed effect with three levels: flat, 2.5cm ridges and 5cm ridges. Seeding was considered as a fixed effect with two levels: seeded and unseeded. Last, month was considered a fixed factor effect with four levels: 1, 3,6 and 12 months. Location was considered a random effect with two levels: Devonport and Westhaven. All univariate analyses were conducted in R-studio.

3.4 Results

Table 3.1: Significance of complexity types upon fouling community biodiversity at highest interaction (NS = no significance, +VE or -VE = significant positive or negative effect, respectively, † = marginal non-significance, M = significant treatment by Month interaction)

Spatial heterogeneity	Cover (%)	Richness	Simpsons	Shannons
Enhancement	M	+VE	-VE	M
Rainfall	NS	NS	NS	NS
Enhancement: Rainfall	NS	+VE	NS	+VE
Habitat utilisation				
Habitat	M	+VE	NS	M
Rainfall	†	+VE	NS	+VE
Enhancement	M	+VE	M	+VE
Habitat: Rainfall	NS	NS	NS	+VE
Enhancement:Habitat	M	+VE	NS	+VE
Enhancement: Rainfall	NS	NS	NS	NS
Enhancement:Rainfall: Habitat	†	+VE	+VE	NS

Table 3.2: Significance of complexity types upon mobile community biodiversity at highest interaction (NS = no significance, +VE or -VE = significant positive or negative effect, respectively, † = marginal non-significance, M = significant treatment by Month interaction)

Spatial heterogeneity	Abundance	Richness	Simpsons	Shannons
Enhancement	+VE	+VE	-VE	+VE
Rainfall	NS	NS	NS	NS
Enhancement: Rainfall	NS	NS	NS	NS
Habitat utilisation				
Habitat	M	+VE	M	+VE
Rainfall	NS	NS	NS	NS
Enhancement	+VE	+VE	NS	+VE
Habitat: Rainfall	NS	NS	NS	NS
Enhancement:Habitat	+VE	NS	NS	M
Enhancement: Rainfall	NS	NS	NS	NS
Enhancement: Rainfall: Habitat	NS	NS	NS	NS

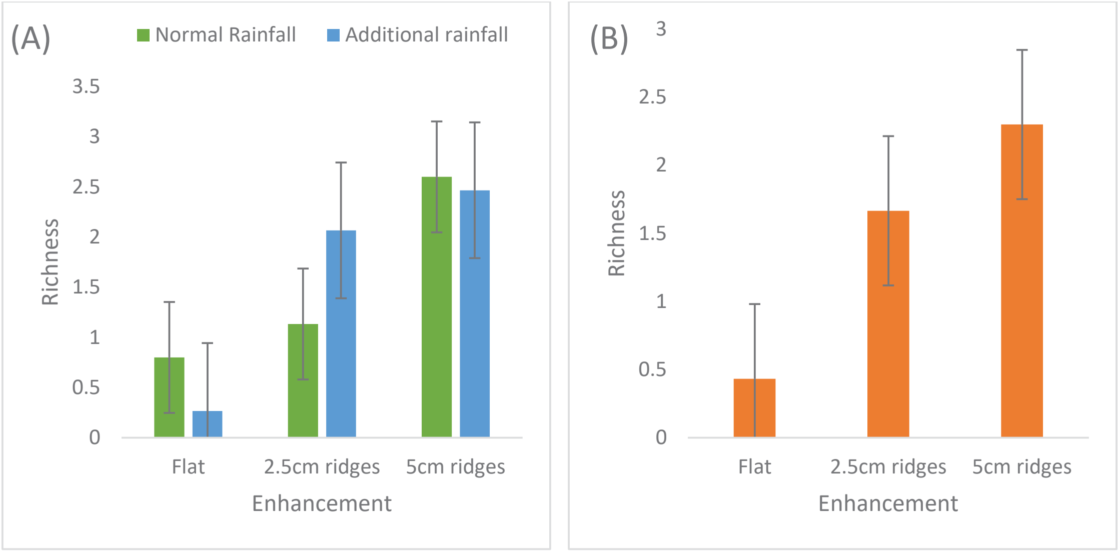

Figure 3.1: Main effects of the enhancement and rainfall on the fouling community's richness (A) as well as the effects of enhancement on the mobile invertebrate's richness (B). All plots denote means ± standard errors.

Additional complexity significantly influenced the mean richness of the fouling (Fig. 3.1A) and mobile (Fig. 3.1B) species positively, interacting with rainfall on the fouling community. The additional rainfall did not however influence the diversity with any discernible pattern across the enhancements.

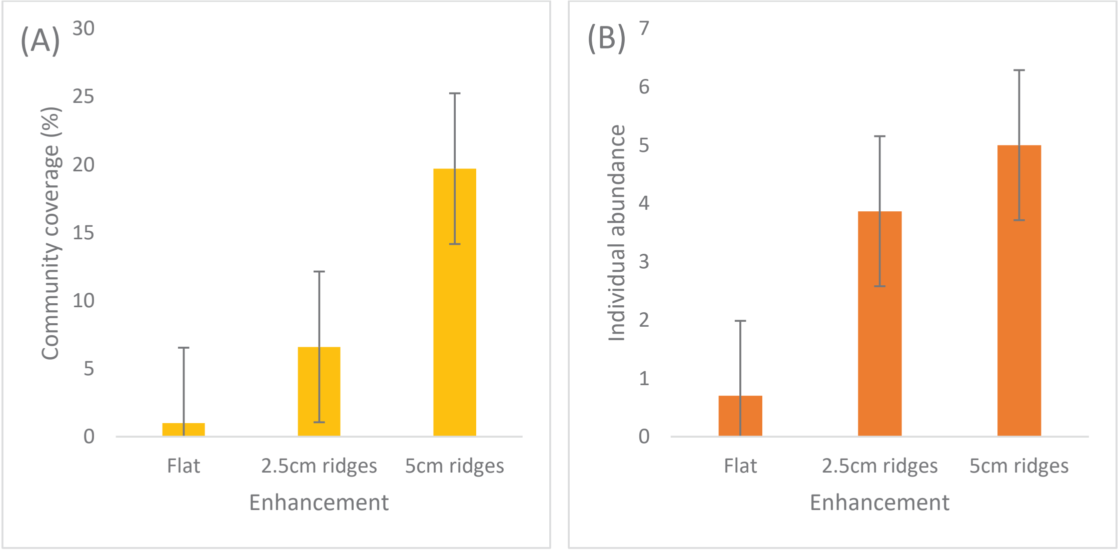

Figure 3.2: Main effects of the enhancement the fouling community coverage (A) as well as the effect of enhancement on mobile invertebrate abundance (B). All plots denote means ± standard errors.

For both the fouling (Fig. 3.2A) and mobile (Fig. 3.2B) communities we found that the complexity increased the mean community abundance, unaffected by additional rainfall.

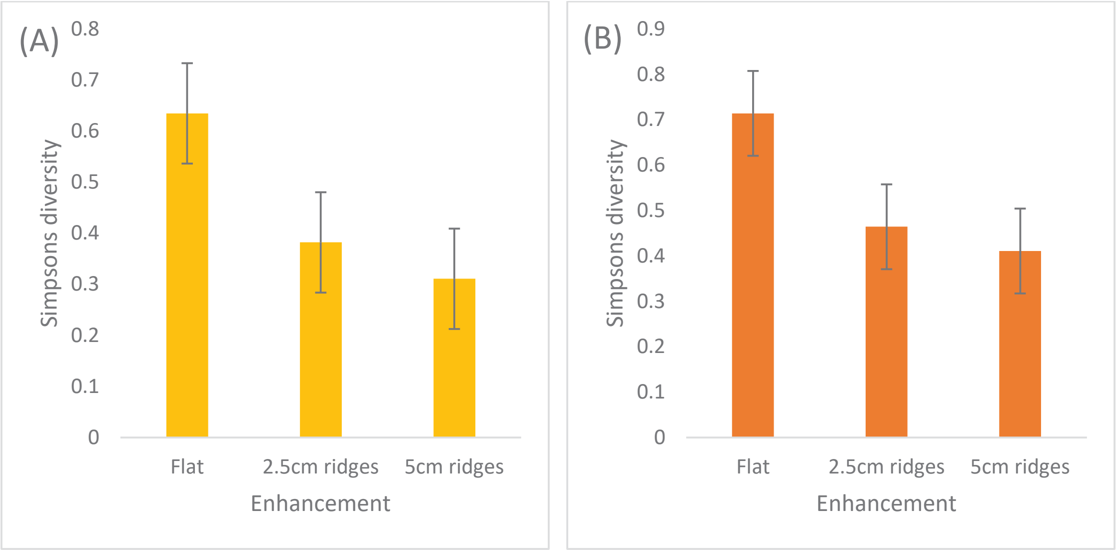

Figure 3.3: Main effects of the enhancement on the fouling community's Simpsons diversity (A) as well as the effect of enhancement on mobile invertebrates Simpsons diversity (B). All plots denote means ± standard errors.

The mean Simpsons diversity of the settlement plates fouling (Fig. 3.3A) and mobile (Fig. 3.3B) communities was found negatively influenced by additional complexity, uninfluenced by additional rainfall.

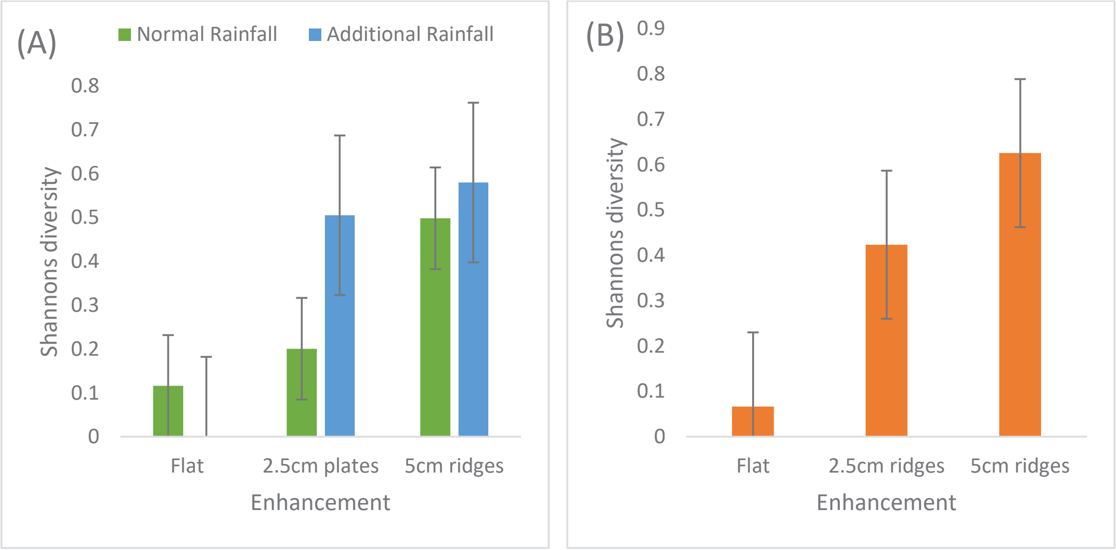

Figure 3.4: Main effects of enhancement and rainfall on the Shannons diversity of the fouling community (A) as well as the effects of enhancement on the mobile invertebrates Shannons diversity (B). All plots denote means ± standard errors.

In both the fouling (Fig. 3.4A) and mobile (Fig. 3.4B) communities, greater complexity increased Shannons diversity. Additional rainfall influenced the fouling community to increase the mean diversity upon the ridged plates.

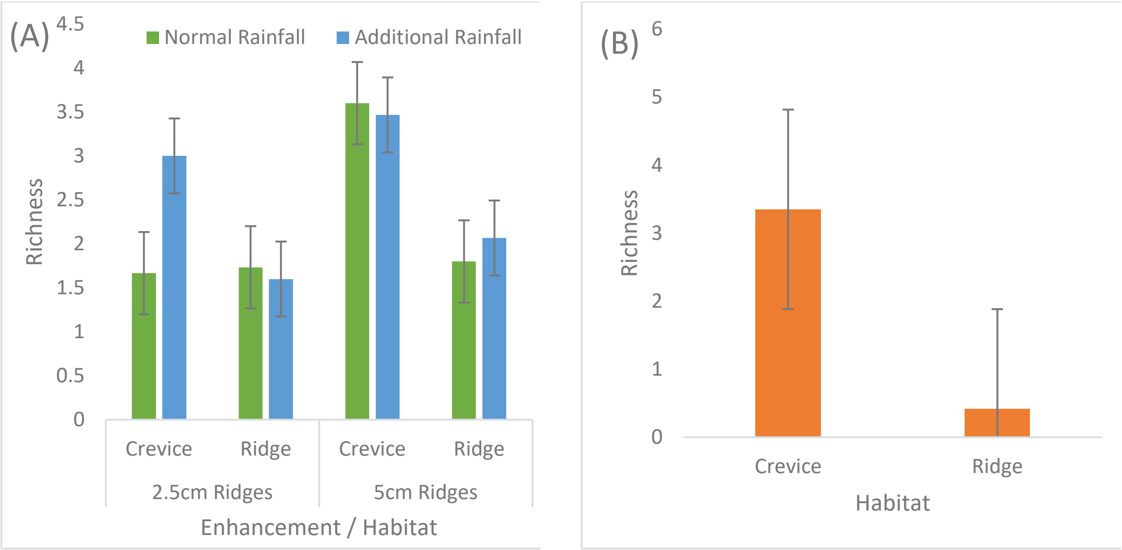

Figure 3.5: Main effects of the enhancement and rainfall on the fouling community richness with habitat utilisation (A) as well as the effects of habitat utilisation on the mobile invertebrate's richness (B). All plots denote means ± standard errors.

The differences between crevice and ridge habitats of the ridged settlement plate significantly affected the biodiversity of the fouling and mobile communities, interacting with rainfall in the fouling community. The fouling community's diversity was influence by interacting effects of habitat, complexity, and rainfall. These influence the species richness (Fig. 3.5A), coverage (Fig. 3.6A), Simpsons diversity (Fig. 3.7A), and Shannons diversity (Fig. 3.8A, 3.8B), generally increasing diversity positively with complexity, additional rainfall, and in the crevices. Rainfall, however, reduced the diversity in the crevices of the 5cm plates of all but Shannons diversity.

The mobile community had a mean increase of richness (Fig. 3.5B), abundance (Fig. 3.6B), and Shannons diversity (Fig. 3.8C) in the crevices, with a greater effect on abundance on the 5cm plates. The mean Simpsons diversity of the mobile community however, was lower in the crevices (Fig. 3.7B).

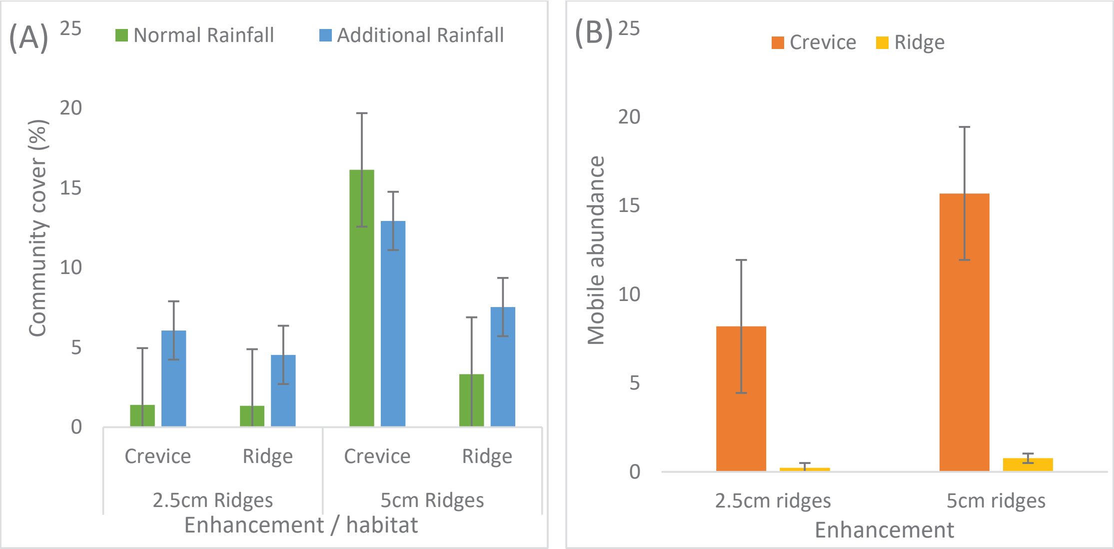

Figure 3.6: Main effects of the enhancement and rainfall on the fouling community coverage with habitat utilisation (A) as well as the interactive effects of enhancement and habitat utilisation on the mobile invertebrate's abundance (B). All plots denote means ± standard errors.

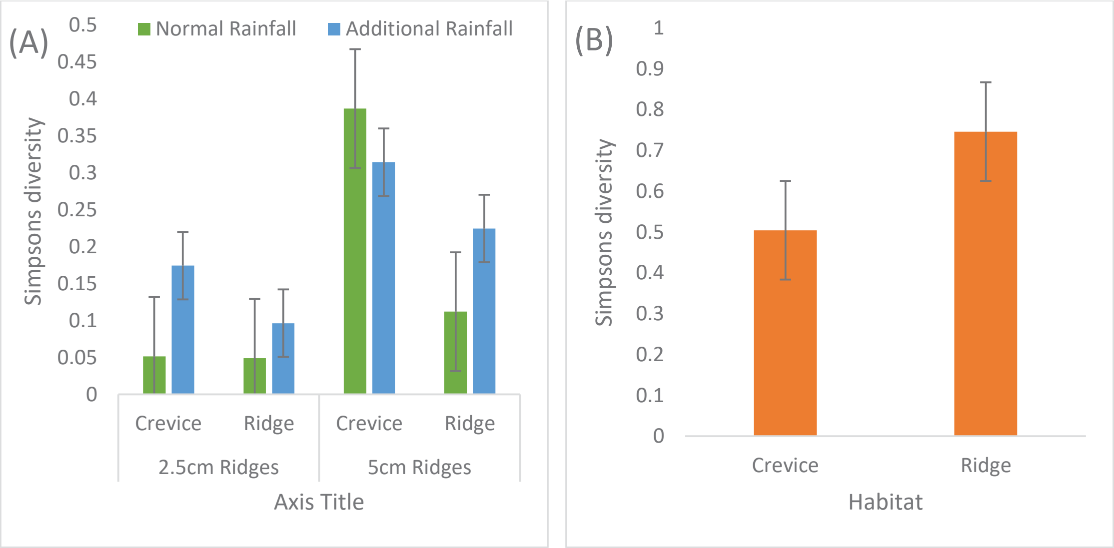

Figure 3.7: Main effects of the enhancement and rainfall on the fouling community richness with habitat utilisation (A) as well as the effects of habitat utilisation on the mobile invertebrates Simpsons diversity (B). All plots denote means ± standard errors.

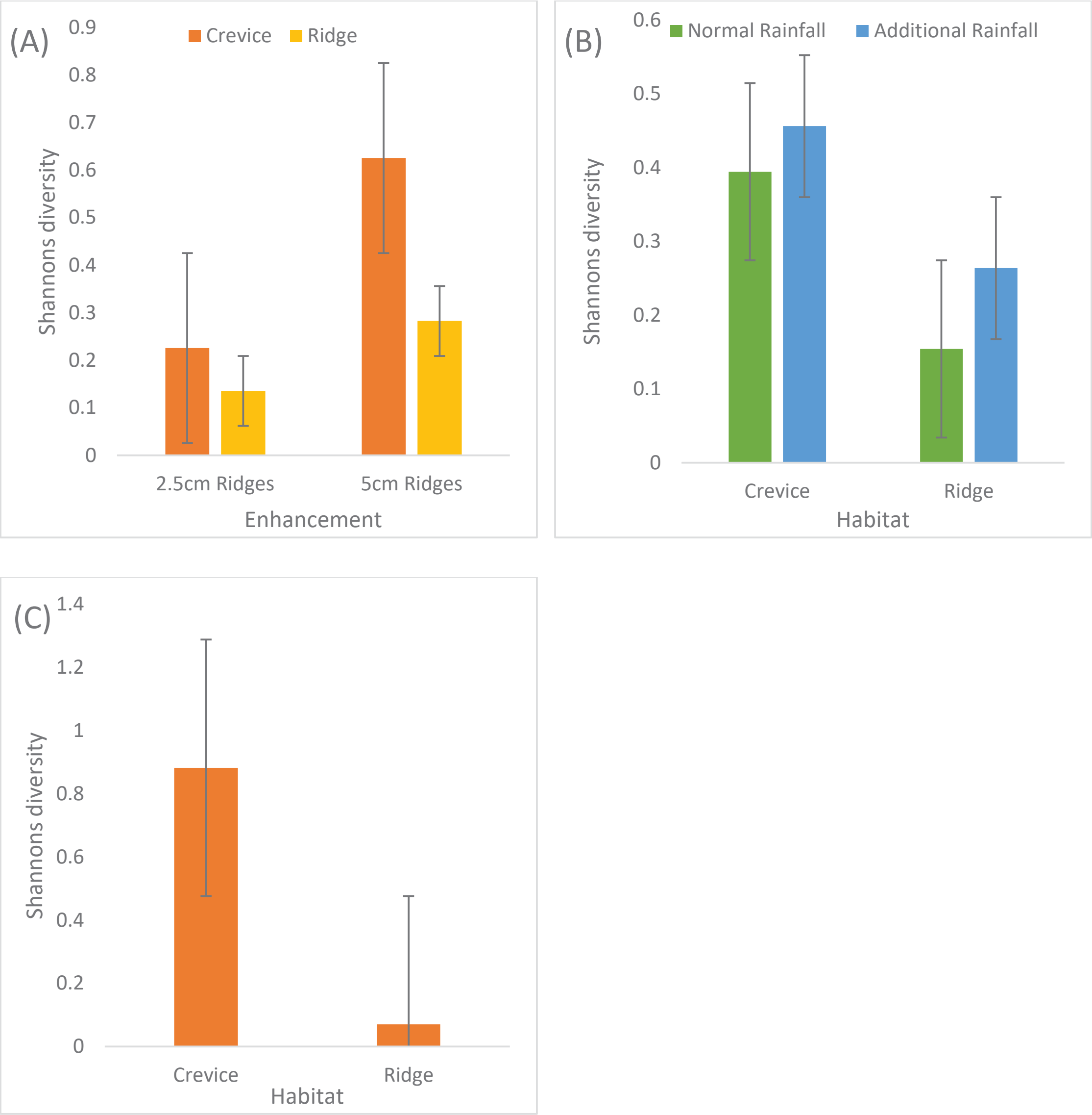

Figure 3.8: Main effects of separate interactions of habitat utilisation with enhancement (A) and rainfall (B) on the Shannons diversity of the fouling community as well as the effect of habitat utilisation on mobile invertebrate species Shannons diversity (C). All plots denote means ± standard errors.

3.5 Discussion

The results provide additional support for the paradigm that additional spatial heterogeneity increases biodiversity. The greatest effects on diversity were observed between the flat plates and ridged plates, supporting that the eco-engineered design, greatly increases diversity over uniform hard-substrates. The basis for this result evident by the habitat utilisation of the communities, with significantly greater diversity, consistently located within the crevices. The addition of these crevices provides shade, water retention, and shelter to the community, relieving thermal and desiccation stress and protecting from breaking waves and predation (Bulleri & Chapman, 2010).

The Simpsons diversity displayed however consistently found opposite effect to complexity and habitat utilisation on diversity. The settlement plates in this experiment developed a very low richness in both the fouling and mobile communities over the length of the experiment, with a mean richness less than three on the most complex plates. The low richness allowed the plates to be dominated by a few species. As Simpsons diversity more heavily weights abundant species relative to rarer species, areas with less species richness, such as the ridges, were classified as having higher diversity. Given the low species richness quantified in this study, Simpsons diversity cannot be used as an accurate representation of the effects on the community. Shannons diversity is perhaps a more appropriate measure of biodiversity, as it favours evenness within the communities, putting more weight on the less abundant species.

The simulation of additional rainfall to the settlement plates had conflicting results. Only the fouling community had any response, with species richness and Shannons diversity significantly influenced by the additional rainfall. The effect however, was not consistent between the complexity levels. Analysing the habitat utilisation, did find a weak consensus that additional rainfall may be increasing the diversity of the fouling community upon ridged plates. This was opposed however in the crevices of the 5cm plates, where diversity was lower on the rained plates. The non-effect of rainfall on the flat plates may result as they have no ability to retain the water like the ridged plates.

This potential relationship of rainfall upon fouling diversity, which was supported within the diverse habitats, may be a consequence of desiccation stresses being relieved by additional rainfall. Therefore, this would allow fouling species, previously restricted to lower sections of seawall, to increase their distribution vertically into higher intertidal heights, providing there is spatial complexity to retain the additional moisture. While there was no effect on the mobile community in our experiment, had the fouling community been able to mature past our three-month timeframe, the distinct diversity of the fouling may become more pronounced and attract additional mobile invertebrate's diversity as a secondary response to the extra rainfall. This would ultimately increase the biodiversity of the higher intertidal heights. However, the additional diversity would increase competition within this environment and would result species being pushed out of their current habitat. While some species may be able to adapt through further vertical distribution changes, others may face extinction if they do not extend their horizonal distribution (Helmuth *et al.,* 2006). If additional rainfall does increase diversity of the intertidal zone, then climate change will increase the spread of NIS through its modification to precipitation patterns. Further research is necessary on the effects precipitation patterns have on intertidal communities to verify the potential effects of climate change.

By and large, our results supported that ecoengineering on artificial seawalls can be used to mitigate the deterioration of diversity resulting from marine anthropogenic modification and that the rocky intertidal zone can be used to monitor climate change effects other than temperature on an ecosystem. The rainfall manipulation suggested that additional rainfall in the intertidal zone, could increase the biodiversity of the fouling community, however, further research is required to understand the extent which rainfall can influence the distribution of intertidal communities. Preforming such an experiment over a longer period and in areas which are more arid that expect a larger increase in rainfall could yield a more significant result by emphasising the influence additional rainfall has upon the intertidal community.

Chapter 4: A comprehensive review of the species inhabiting the Waitematā Harbour.

Photo 3.1: Common Waitematā fish species *Chrysophrys auratus* from Goat Island in the northern Auckland region. Photo credit: Jenny Enderby, retrieved from http://www.stuff.co.nz/auckland/local-news/rodney-times

4.1 Abstract

Despite numerous independent surveys of the Waitematā Harbour, there is yet to be a complete species list for the harbour community. This literature reviews the available literature to provide a unified list of the species found within the Waitematā Harbour. Some species “known” to occur in the Harbour will be missing from the list, as they have not been explicitly recorded in the harbour in the published literature. The summery of the community composition found an alarming number of NIS identified within the harbour. The majority of the NIS were fouling species and it is speculated that the lack of habitat complexity on artificial structures with the harbour is allowing a greater rate of fouling species to invade.

4.2 Introduction

The city of Auckland is framed by two harbours, the Manukau harbour to the south and the Waitematā harbour to the north. However, it is the smaller, northern, Waitematā Harbour (36^{O}50’ S, 174^{O}45’ E), which is often referred to as “Auckland harbour”, as it is the main access and shipping route for the city of Auckland. The harbour’s deep channels and protection from wind and swell played a significant role in the formation of the city of Auckland and in making the Ports of Auckland New Zealand’s busiest port.

The Waitematā Harbour is the remnant of a drowned river valley that was formed in the early Miocene epoch around 23 million years ago through the volcanism (Searle, 1981). Following this period, erosion of these volcanos and deposited material from New Zealand’s Northland Allochthon created the sandstone and mudstone that are widespread throughout the area (Edbrooke, 2001). Rising sea levels created the harbour and since then the coast has been continually reshaped through volcanism. The greatest example being that the harbour once had two entrances around the Devonport volcanos, until and the northern entrance was closed by the Pupuke eruption (Searle, 1981). The current Harbour entrance between North head in the north and Bastion point in the south, opens into the Hauraki Gulf behind Rangitoto, Motutapu and Waiheke islands, which further serve as a natural wave break for the harbour (Dromgoole & Foster, 1983).

The harbour area is over 180km^2, has a maximum depth of 27m at the harbour entrance and extends west from the Harbour entrance for 18 kilometres (Waterhouse, 1966). The harbour’s substrate is primarily soft-mud except for the main channel within the mid harbour which has a coarse shell substratum (Powell, 1937). Unmodified coastlines consist of tidal mudflats, predominantly in the north and west, saltmarsh in the southwest, scattered rocky shores and a small number of sandy beaches in the mid harbour. The Auckland Harbour Bridge crosses the Harbour at its narrowest point, between Westhaven and Northcote Point, and is commonly used to distinguish the two sections of the harbour (Hounsell, 1935).

The Upper Harbour includes the areas west of the Harbour Bridge. This section of the Harbour is dominated by tidal mudflats and saltmarsh and although direct anthropogenic modification is minimal, the Upper Harbour is susceptible to indirect modification through polluted storm-water discharge, sedimentation, and nutrient overloading from agricultural runoff (Hewitt *et al.*, 2006). Accordingly, the ecology of the Upper Harbour is strongly influenced by the rivers that feed into the harbour from an urban catchment to the south and west and a larger rural catchment to the northwest.

The Mid Harbour is the area extending east from the Harbour Bridge to the harbour entrance. The southern coastline of the Mid Harbour has been almost completely replaced with artificial substrates, including the Ports of Auckland, New Zealand's largest marina at Westhaven, and the Auckland central business district's (CBD) waterfront. The catchment of the Mid Harbour is almost exclusively urban, and is susceptible to pollution from storm discharge, overflowing sewage systems, as well as activities associates with the six marinas, the Ports of Auckland, and the Devonport Naval Base which access the Mid Harbour.

In addition to the natural shelter provided by the local geography, the Waitematā lacks shoals or sandbars and is deep enough to berth large ships close to shore on a lowland shoreline. These beneficial conditions facilitated trade for indigenous Māori communities and European settlers, resulting in Auckland rapidly developing into New Zealand's largest settlement (Stone, 2002). Accordingly, the Waitematā Harbour is one of the earliest and most heavily human modified marine ecosystems in New Zealand.

The earliest large-scale marine modification of Waitematā Harbour occurred around 1860, when shores near the current Auckland CBD were reclaimed as wharves and breakwaters during construction for the Ports of Auckland (Dromgoole & Foster, 1983). Nowadays, the Ports of Auckland are not the only notable port within the Waitematā, with the Devonport Naval Base, the Kauri Point Armament Depot, and the Chelsea Sugar Refinery all capable of berthing ships over 500 gross registered tons. The Waitematā also contains seven marinas, including New Zealand's largest at Westhaven, nine ferry terminals, and is scattered with smaller wharves, buoys, moorings, and seawalls across its coastline.

By the end of the 20th century, the Ports of Auckland alone provided international shipping links to 207 foreign ports in 73 other countries (www.arc.govt.nz) and as New Zealand's largest port, handles 43% of New Zealand's total container trade, processing 68% of the country's imports and 33% of its exports by value (www.poal.co.nz). By virtue of being the epicentre of the countries marine international connections, it is also a concerning pathway for the introduction of marine non-indigenous species (NIS) into the New Zealand communities.

4.2.1 Taxonomic accounts

The Waitematā harbour ecosystem supports a diverse array of temperate estuarine and coastal communities, inhabiting soft sediments, rocky reefs, tidal mudflats and salt marshes. The original taxonomic accounts of New Zealand's marine biota have been speculated to have been collected from Auckland, including the Waitematā, however the locality of the specimens was never specified (Dromgoole & Foster, 1983). Some the earliest taxonomic descriptions of species inhabiting the Waitematā come from Suter H. (1907) who collected New Zealand Notoacmea. The first marine biotic survey within the harbour was performed by Oliver (1923), who examined intertidal communities at Westmere Reef as a case study in his wider account of New Zealand communities. In 1937, Powell was the first to perform a large-scale survey of the Waitematā benthic geology and biological communities, providing the earliest descriptions of the ecosystem attributes and general ecological status. As Auckland has developed, the Auckland City Council has shown an increased interest in the harbours condition, performing six consecutive years of ecological monitoring within the Mid Harbour from 2000 (Halliday *et al.*), a benthic survey of the Upper Harbour in 2002 (Hewitt *el al.*), and annual surveys of non-indigenous species within the Ports of Auckland since 2006 (Inglis *et al.*).

The benthic communities of the Waitematā have been repeatedly monitored since Powell's original survey in 1937, however the systems taxonomy has never been collated into a single document. The complete taxonomy of the harbour still required attention, as many surveys focus on particular facets of the system and therefore each represent a fraction of the total taxonomy. This has resulted in some marine species to be continually overlooked within certified publications. Many such species are known to occur within the surrounding Hauraki gulf, however as they have not been explicitly identified within the harbour, they were excluded from the species list. The best described communities within the Waitematā include the saltmarshes and mudflats (Chapman & Ronaldson, 1958), Foranminifera (Hayward *et al*., 1997_1), and Ostracoda (Morley & Hayward, 2007). Overall, despite numerous biological surveys and studies on individual taxa, the literature on the biota of the Waitematā Harbour is scattered and much of the taxonomy requires updating and confirmation, thus this checklist is an attempt to address those needs.

4.3 Methods

The species list presented here includes only those taxa that have been identified in the area west of the Harbour entrance between North Head and Bastion Point and the lower reaches of the brackish tidal rivers of the upper harbour. The results were limited to research papers and book chapters which could be accessed through the search engines: Discover, Scopus, Web of Science, and Google Scholar. Searching terms always included "Auckland", "Waitematā" and "marine", as well as a combination of the terms: "taxa", "taxonomy", "species", "biolog*", "ecolog*", "introduction", "survey", "study", "analysis", "invasive", and "indigenous". As literature was reviewed, we expanded our search to find the original citations for each taxon. The literature which possessed the earliest positive identification of a species was used as the reference. Accordingly, species listed in articles appearing in grey literature which have not been cited may have been missed. Widespread species that may be expected to occur within the Waitematā Harbour, but have not been explicitly recorded within the harbour have not been included. Systematics and synonymy follow the primary taxonomic literature wherever possible, referring to the World Register of Marine Species (WoRMS; http://www.marinespecies.org/).

4.4 Results

A total of 546 taxa from 285 families have been recorded with the Waitematā Harbour (Appendix. 3.2), within 61 publications (Appendix. 3.1). Macroalgae (6.2%), Porifera (4.2%), Polychaeta (9.9%), Mollusca (19.4%), Crustacea (17.7%), Bryozoa (5.9%), Ascidians (5.1%), fish (6%), and birds (3.1%) made up 78% of the reported species (Table 4.1)

Table 4.1. Marine taxonomy composition recorded within the Waitematā Harbour

	Families	Taxa	Endemic	% Endemic	NIS	% NIS
Foraminifera	31	63	8	12.7%	1	1.6%
Myzozoa	2	6	0	0.0%	0	0.0%
Bigyra	1	1	1	100.0%	0	0.0%
Phaeophyceae	8	15	4	26.7%	4	26.7%
Cholorophyta	4	8	0	0.0%	1	12.5%
Rhodophyta	9	11	0	0.0%	1	9.1%
Tracheophyta	9	13	3	23.1%	3	23.1%
Porifera	11	23	5	21.7%	9	39.1%
Cnidaria	11	14	0	0.0%	10	71.4%
Polychaeta	25	54	30	55.6%	9	16.7%
Mollusca	52	114	92	80.7%	14	12.3%
Chelicerata	1	1	0	0.0%	0	0.0%
Crustacea	54	97	52	53.6%	13	13.4%
Bryozoa	18	32	5	15.6%	21	65.6%
Brachiopoda	1	1	1	100.0%	0	0.0%
Echinodermata	8	12	7	58.3%	0	0.0%
Hemichordata	1	1	0	0.0%	0	0.0%
Tunicata	8	28	9	32.1%	13	46.4%
Pisces	21	33	11	33.3%	3	9.1%
Aves	9	17	6	35.3%	0	0.0%
Mammalia	1	2	0	0.0%	0	0.0%
Total	285	546	212	38.8%	102	18.7%

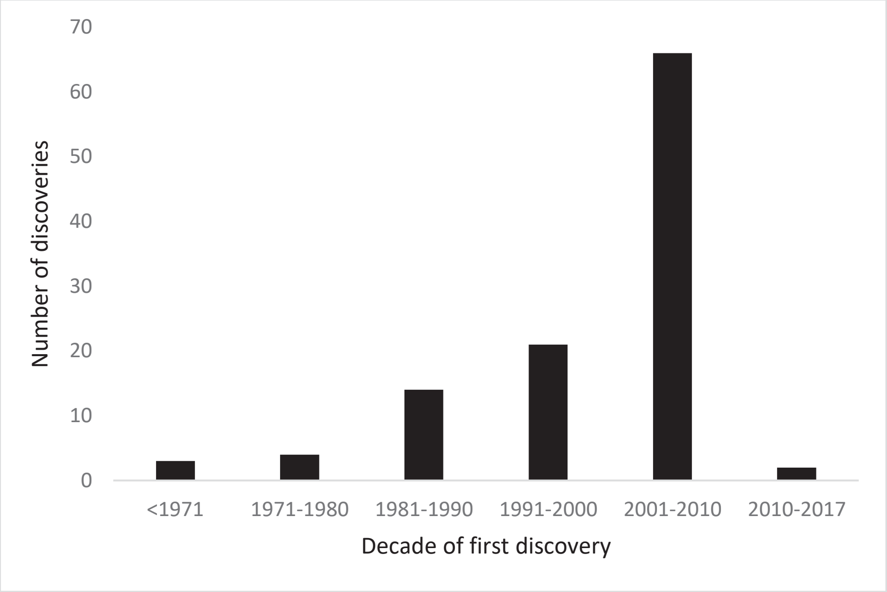

Figure 4.1. Decade of discovery of NIS within the Waitematā Harbour

Figure 4.1. indicates the decade in which the NIS were identified within the Waitematā Harbour, which appears to show an exponential growth in the rate of species discovery. However, it is more likely that many of the species have been present before their discovery and it is only due to increased research into the ecosystem that the species were identified. This has resulted in a rise in discoveries at the end of the 20th century which peaked between 2001-2010 when the Auckland Regional Council issued three major studies within the Upper Harbour, Mid Harbour, and Ports of Auckland. Since then, as most of the NIS have been identified within the system, only new introductions or species which these surveys could have missed can be reported, resulting in the sudden drop in discoveries in the current decade.

In total, 102 species which comprise nearly 19% of all taxa identified within Waitematā Harbour are classified as non-indigenous species (NIS). Of these NIS, Bryozoans are the most common NIS contributing to an alarming 65.6% of the Bryozoan fauna within the Harbour. Molluscs and Crustaceans are heavily represented in the NIS community, but they only made up 12.3% and 13.4 % of each fauna respectively. Ascidians and Cnidarians follow in NIS richness but contribute to 46.4% and 71.4% of the faunal richness due to their lower total richness (Table 1.). Overall, sessile species accommodate the majority of the NIS identified within the Waitematā Harbour, with the heaviest detriment to the local fouling bryozoan, ascidian, porifera, and cnidarian communities.

4.5 Discussion

The abundance of NIS identified within this review emphasises the detrimental effects that marine modification is having the local communities. Nearly a fifth of the confirmed species within the Waitematā were non-indigenous to New Zealand waters. Of these, local sessile communities are being over taken in richness by NIS, as the low local richness is overwhelmed by the ratio of invasive within the following communities; Porifera (39%), Ascidians (46%), Bryozoans (66 %), and Cnidarians (71%). Such invaders usually colonise hard substrate environments, such as artificial modification. Their abundant success as invaders can therefore suggest that artificial modifications are facilitating their successful establishment into the Waitematā.

Hard substrate fouling communities have been suggested as the easiest areas which marine NIS are able to invade (Bumbeer & Rocha, 2012). To a degree, this is a combination resulting from modifying the coastlines into artificial hard substrate, urban pollution lowering local community dominance, and as sessile fouling species can easily spread through free swimming life stages (Piola & Johnston, 2008: Megina *et al*., 2013). NIS generally have a high tolerance for heavy metal pollutants which occur in high concentrations within areas such as ports and marinas, while local fouling fauna have been found to be reduced by up to 40% in modified areas (Crooks *et al.,* 2011). This allows the relatively unaffected fouling NIS from the undersides or ballast water of vessels to invade into the communities within international ports highly effectively. With the global modification of harbour, ports, and seawalls and the ability for NIS to tolerate thermal conditions beyond their local range (Lewis *et al.* 2006), the fouling hard substrate communities are at risk of becoming homogenous worldwide. As invasive fouling communities naturalise within a harbour, mobile NIS that associate with these non-indigenous communties would have a greater ability to invade.

This effect of previous invasions facilitating the introduction of more NIS is also found to be the result of physiological traits of fouling NIS. Fouling NIS have been found to have negative effects on the post-fertilisation performance of the local sessile species, inhibiting the settlement of native larvae around themselves and increasing their post settlement mortality tenfold (Rius *et al.* 2009). Therefore, the susceptibility of hard substrate communities for invasion may be increasing with each new NIS introduction, particularly within polluted areas where the detrimental effects accumulate upon the local community (Piola & Johnston, 2008), allowing a potential growing rate of successful invasions over time.

5.1 Chapter summaries and overall synthesis

The results from the experiments in chapter two confirmed the paradigm that habitat heterogeneity increases biodiversity. Moreover, we found that biodiversity was positively influenced by both physical habitat complexity and biological complexity. The greatest benefits of habitat complexity for biodiversity appeared driven by the presence of crevices. The crevices provided the fouling and mobile communities with shade, greater water retention, and less exposure to swells and thereby provided relief from thermal and desiccation stresses and reduced potential for dislodgement and predation. The biological complexity contributed by the seeded mussels independently introduced additional crevices between the plate and their shell which provided a variety of crevice sizes. While the benefits of crevices generated by the seeded mussels for biodiversity measured in our experiment would not be as pronounced within a dense natural population of mussels, locations with sparse mussel populations would benefit greatly from this additional habitat complexity. The existing seawalls in the locations where the experiments were deployed, accumulated biodiversity more rapidly and achieved a greater biodiversity than the uniform, flat settlement plates. However, despite the ability of the existing seawall to rapidly regenerate their fouling community, the surface did not produce equivalent diversity to the ridged, eco-engineered settlement plates.

Chapter three showed that climate change driven increases in rainfall are unlikely to drive major changes in the biodiversity and community composition of intertidal communities. The additional rainfall did not have any significant effect on the mobile invertebrate community. There was however, a suggestion that the fouling community may become more diverse with additional rainfall, indicating that increased rainfall could reduce the stress from desiccation, allowing fouling communities to shift their distribution vertically, from the lower intertidal. Given enough time for the fouling community to fully develop, the mobile communities might also respond to changes in the fouling community as a secondary response to the increased rainfall though we did not explicitly examine such an effect. A vertical shift by species typically found in the lower intertidal may force cascading shifts in the distributions of mid intertidal species (Helmuth *et al.,* 2006). It is therefore, possible that climate change could increase the spread of NIS through influencing precipitation patterns.

Further research is necessary on the effects precipitation patterns have on intertidal communities to verify the potential effects of climate change.

The literature review presented in chapter four indicated that the Waitematā Harbour has been heavily influenced by anthropogenic disturbances –nearly 20% of the species identified were non-indigenous. Of these NIS, porifera, ascidians, bryozoans, and cnidarians alone make up over 50% of the NIS pool and a significant portion of the species richness for each of these phyla is NIS. Moreover, many of the species within these phyla are members of the fouling community and it is likely that artificial structures are facilitating their successful establishment into the Waitematā. Therefore, to mitigate the largest invasion threat to the Waitematā Harbour and port across the globe, ecoengineered seawalls appear as a practical solution to limit the successful establishment of fouling NIS into local communities.

Foreshore modification is known to greatly reduce the local biodiversity of intertidal communities (Glasby, 1999). This is chiefly a result of previous designs lacking spatial heterogeneity, which increases the diversity of available habitats and allows a more diverse community to exist (Levin, 1992). The key outcome of our research indicates that implementing ecoengineering into the design and construction of future marine structures, will bolster the local biodiversity greatly over uniform designs. Consequentially, the greater biotic resistance of a diverse community of native species will mitigate the ability of NIS to establish in new bioregions (Ferreira, 2003). Hard-substrate environments have been proposed as the easiest environment for marine NIS to establish (Bumbeer & Rocha, 2012), which is supported by the results of the Waitematā Harbour species composition, which is primarily afflicted with fouling NIS.

For future research, I would suggest the design of settlement plates that continue to utilise greater forms of habitat complexity. I would like to see if there is an optimal width and depth for the crevices to increase intertidal diversity or if the implementation of more variable crevice sizes on a settlement plate will produce more diversity than any single width and depth. Alternatively, I would have liked to have seen if the orientation of the settlement plates affect diversity, as horizontal ridges would have greater water retention. I would have liked to continue the rainfall experiment, to confirm if the mature fouling community is influenced by additional rainfall. Additionally, if the rainfall manipulation was performed in a more arid environment or in an area which expected a greater rainfall increase, the emphasised influence would yield a more secure result.

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Appendices

Appendix: 1.1: Species richness of the fouling communities.

	Df	Sum Sq	Mean Sq	F value	Pr(>F)
Enhancement	2	223.910	111.955	66.567	0.015*
Seeding	1	58.770	58.770	3395.051	0.011*
Site	1	71.289	71.289	36.810	0.000*
Month	4	321.710	80.428	13.895	0.013*
Enhancement:Seeding	2	17.751	8.876	11.066	0.083
Enhancement:Site	2	3.364	1.682	0.868	0.421
Enhancement:Month	8	27.311	3.414	2.038	0.167
Seeding:Site	1	0.017	0.017	0.009	0.925
Seeding:Month	4	0.765	0.191	0.058	0.991
Site:Month	4	23.152	5.788	2.989	0.020*
Enhancement:Seeding:Site	2	1.604	0.802	0.414	0.661
Enhancement:Seeding:Month	8	13.194	1.649	0.603	0.755
Enhancement:Site:Month	8	13.400	1.675	0.865	0.547
Seeding:Site:Month	4	13.122	3.280	1.694	0.152
Enhancement:Seeding:Site:Month	8	21.887	2.736	1.413	0.192
Residuals	225	435.750	1.937		

Appendix: 1.2: Species richness of the mobile invertebrates.

	Df	Sum Sq	Mean Sq	F value	Pr(>F)
Enhancement	2	124.087	62.043	6.557	0.132
Seeding	1	66.435	66.435	977.875	0.020*
Site	1	6.292	6.292	7.251	0.008*
Month	4	222.093	55.523	17.890	0.008*
Enhancement:Seeding	2	14.217	7.108	15.848	0.059†
Enhancement:Site	2	18.923	9.462	10.903	0.000*
Enhancement:Month	8	22.240	2.780	2.647	0.095
Seeding:Site	1	0.068	0.068	0.078	0.780
Seeding:Month	4	8.586	2.146	2.962	0.159
Site:Month	4	12.414	3.104	3.577	0.008*
Enhancement:Seeding:Site	2	0.897	0.449	0.517	0.597
Enhancement:Seeding:Month	8	2.885	0.361	0.762	0.645
Enhancement:Site:Month	8	8.401	1.050	1.210	0.294
Seeding:Site:Month	4	2.899	0.725	0.835	0.504
Enhancement:Seeding:Site:Month	8	3.788	0.474	0.546	0.821
Residuals	225	195.250	0.868		

Appendix: 1.3: Fouling coverage (%) across the settlement plates.

	Df	Sum Sq	Mean Sq	F value	Pr(>F)
Enhancement	2	804.860	402.430	0.480	0.676
Seeding	1	350.342	350.342	0.454	0.623
Site	1	3527.563	3527.563	16.915	0.000*
Month	4	61316.848	15329.212	5.777	0.059
Enhancement:Seeding	2	94.829	47.414	0.165	0.858
Enhancement:Site	2	1676.490	838.245	4.020	0.019*
Enhancement:Month	8	730.928	91.366	0.276	0.956
Seeding:Site	1	772.229	772.229	3.703	0.056 †
Seeding:Month	4	2697.477	674.369	1.128	0.455
Site:Month	4	10613.583	2653.396	12.723	0.000*
Enhancement:Seeding:Site	2	574.937	287.468	1.378	0.306
Enhancement:Seeding:Month	8	1503.322	187.915	0.744	0.516
Enhancement:Site:Month	8	2645.229	330.654	1.586	0.130
Seeding:Site:Month	4	2392.156	598.039	2.868	0.024*
Enhancement:Seeding:Site:Month	8	2020.553	252.569	1.211	0.293
Residuals	225	46922.200	208.543		

Appendix: 1.4: Mobile invertebrate abundance across the settlement plates.

	Df	Sum Sq	Mean Sq	F value	Pr(>F)
Enhancement	2	9021.994	4510.997	11.731	0.079
Seeding	1	10645.639	10645.639	29.900	0.115
Site	1	2359.327	2359.327	30.303	0.000*
Month	4	30073.980	7518.495	14.447	0.012*
Enhancement:Seeding	2	62.144	31.072	0.098	0.910
Enhancement:Site	2	769.053	384.527	4.939	0.008*
Enhancement:Month	8	3246.511	405.814	1.418	0.317
Seeding:Site	1	356.036	356.036	4.573	0.034*
Seeding:Month	4	6860.915	1715.229	18.555	0.008*
Site:Month	4	2081.738	520.435	6.684	0.000*
Enhancement:Seeding:Site	2	631.887	315.943	4.058	0.019*
Enhancement:Seeding:Month	8	151.529	18.941	0.365	0.912
Enhancement:Site:Month	8	2290.261	286.283	3.677	0.000*
Seeding:Site:Month	4	369.767	92.442	1.187	0.317
Enhancement:Seeding:Site:Month	8	415.421	51.928	0.667	0.720
Residuals	225	17518.100	77.858		

Appendix: 1.5: Fouling communities Simpsons diversity.

	Df	Sum Sq	Mean Sq	F value	Pr(>F)
Enhancement	2	2.668	1.334	7.263	0.121
Seeding	1	1.335	1.335	15.411	0.159
Site	1	0.035	0.035	1.291	0.257
Month	4	0.450	0.112	1.615	0.327
Enhancement:Seeding	2	0.257	0.129	0.988	0.503
Enhancement:Site	2	0.367	0.184	6.840	0.001*
Enhancement:Month	8	0.296	0.037	3.157	0.062
Seeding:Site	1	0.087	0.087	3.228	0.074
Seeding:Month	4	0.087	0.022	0.943	0.522
Site:Month	4	0.279	0.070	2.594	0.037*
Enhancement:Seeding:Site	2	0.261	0.130	4.852	0.009*
Enhancement:Seeding:Month	8	0.289	0.036	1.343	0.343
Enhancement:Site:Month	8	0.094	0.012	0.437	0.898
Seeding:Site:Month	4	0.092	0.023	0.856	0.491
Enhancement:Seeding:Site:Month	8	0.215	0.027	1.003	0.435
Residuals	225	6.040	0.027		

Appendix: 1.6: Mobile invertebrates Simpsons diversity.

	Df	Sum Sq	Mean Sq	F value	Pr(>F)
Enhancement	2	0.093	0.046	0.316	0.760
Seeding	1	0.114	0.114	0.848	0.526
Site	1	0.634	0.634	7.953	0.005*
Month	4	1.390	0.348	3.493	0.127
Enhancement:Seeding	2	0.199	0.099	0.948	0.513
Enhancement:Site	2	0.294	0.147	1.843	0.161
Enhancement:Month	8	1.523	0.190	6.399	0.008*
Seeding:Site	1	0.134	0.134	1.680	0.196
Seeding:Month	4	1.264	0.316	7.081	0.042*
Site:Month	4	0.398	0.099	1.249	0.291
Enhancement:Seeding:Site	2	0.210	0.105	1.317	0.270
Enhancement:Seeding:Month	8	1.449	0.181	2.861	0.079
Enhancement:Site:Month	8	0.238	0.030	0.373	0.934
Seeding:Site:Month	4	0.179	0.045	0.560	0.692
Enhancement:Seeding:Site:Month	8	0.507	0.063	0.795	0.608
Residuals	225	17.928	0.080		

Appendix: 1.7: Fouling communities Shannons diversity.

	Df	Sum Sq	Mean Sq	F value	Pr(>F)
Enhancement	2	14.098	7.049	20.148	0.047*
Seeding	1	7.036	7.036	115.633	0.059†
Site	1	0.594	0.594	5.942	0.016*
Month	4	4.521	1.130	3.056	0.152
Enhancement:Seeding	2	1.276	0.638	3.647	0.215
Enhancement:Site	2	0.700	0.350	3.498	0.032*
Enhancement:Month	8	1.101	0.138	6.023	0.010*
Seeding:Site	1	0.061	0.061	0.608	0.436
Seeding:Month	4	0.113	0.028	0.224	0.912
Site:Month	4	1.479	0.370	3.698	0.006*
Enhancement:Seeding:Site	2	0.350	0.175	1.749	0.176
Enhancement:Seeding:Month	8	1.113	0.139	1.065	0.466
Enhancement:Site:Month	8	0.183	0.023	0.228	0.985
Seeding:Site:Month	4	0.502	0.125	1.254	0.289
Enhancement:Seeding:Site:Month	8	1.046	0.131	1.307	0.241
Residuals	225	22.503	0.100		

Appendix: 1.8: Mobile invertebrates Shannons diversity.

	Df	Sum Sq	Mean Sq	F value	Pr(>F)
Enhancement	2	7.626	3.813	7.230	0.122
Seeding	1	3.233	3.233	249.375	0.040*
Site	1	0.202	0.202	2.090	0.150
Month	4	19.547	4.887	26.769	0.004*
Enhancement:Seeding	2	2.548	1.274	2596.881	0.000*
Enhancement:Site	2	1.055	0.527	5.460	0.005*
Enhancement:Month	8	1.012	0.126	0.824	0.605
Seeding:Site	1	0.013	0.013	0.134	0.714
Seeding:Month	4	0.958	0.239	1.760	0.299
Site:Month	4	0.730	0.183	1.890	0.113
Enhancement:Seeding:Site	2	0.001	0.000	0.005	0.995
Enhancement:Seeding:Month	8	1.252	0.156	2.736	0.088
Enhancement:Site:Month	8	1.228	0.153	1.589	0.129
Seeding:Site:Month	4	0.544	0.136	1.408	0.232
Enhancement:Seeding:Site:Month	8	0.457	0.057	0.592	0.784
Residuals	225	21.735	0.097		

Appendix: 1.9: Fouling richness with habitat utilisation.

	Df	Sum Sq	Mean Sq	F value	Pr(>F)
Enhancement	1	8.714	8.714	11.539	0.182
Seeding	1	18.438	18.438	22.120	0.133
Location	1	175.743	175.743	103.215	0.062†
Site	1	0.245	0.245	0.130	0.719
Enhancement:Seeding	1	3.618	3.618	1.810	0.407
Enhancement:Location	1	2.396	2.396	3.239	0.323
Enhancement:Site	1	0.755	0.755	0.402	0.527
Seeding:Location	1	0.142	0.142	31.393	0.112
Seeding:Site	1	0.834	0.834	0.443	0.506
Location:Site	1	1.703	1.703	0.905	0.342
Enhancement:Seeding:Location	1	3.857	3.857	3.287	0.321
Enhancement:Seeding:Site	1	1.999	1.999	1.063	0.303
Enhancement:Location:Site	1	0.740	0.740	0.393	0.531
Seeding:Location:Site	1	0.005	0.005	0.002	0.961
Enhancement:Seeding:Location:Site	1	1.173	1.173	0.624	0.430
Residuals	354	665.772	1.881		

Appendix: 1.10: Mobile invertebrate richness with habitat utilisation.

	Df	Sum Sq	Mean Sq	F value	Pr(>F)
Enhancement	1	2.852	2.852	0.240	0.710
Seeding	1	40.564	40.564	17060.469	0.005*
Location	1	254.727	254.727	297.200	0.037*
Site	1	0.391	0.391	0.274	0.601
Enhancement:Seeding	1	2.323	2.323	1.584	0.427
Enhancement:Location	1	2.431	2.431	5.000	0.268
Enhancement:Site	1	11.882	11.882	8.332	0.004*
Seeding:Location	1	2.096	2.096	7.284	0.226
Seeding:Site	1	0.002	0.002	0.002	0.967
Location:Site	1	0.857	0.857	0.601	0.439
Enhancement:Seeding:Location	1	0.033	0.033	0.034	0.884
Enhancement:Seeding:Site	1	1.466	1.466	1.028	0.311
Enhancement:Location:Site	1	0.486	0.486	0.341	0.560
Seeding:Location:Site	1	0.288	0.288	0.202	0.654
Enhancement:Seeding:Location:Site	1	0.975	0.975	0.683	0.409
Residuals	354	504.846	1.426		

Appendix: 1.11: Mobile invertebrate abundance with habitat utilisation.

	Df	Sum Sq	Mean Sq	F value	Pr(>F)
Enhancement	1	1209.673	1209.673	9.125	0.204
Seeding	1	3006.585	3006.585	1306.121	0.018*
Location	1	16582.511	16582.511	25825.030	0.004*
Site	1	331.716	331.716	3.552	0.060†
Enhancement:Seeding	1	0.026	0.026	0.000	0.988
Enhancement:Location	1	1092.997	1092.997	9.728	0.198
Enhancement:Site	1	132.562	132.562	1.419	0.234
Seeding:Location	1	544.735	544.735	10.145	0.194
Seeding:Site	1	2.302	2.302	0.025	0.875
Location:Site	1	0.642	0.642	0.007	0.934
Enhancement:Seeding:Location	1	40.185	40.185	16.735	0.153
Enhancement:Seeding:Site	1	78.304	78.304	0.838	0.360
Enhancement:Location:Site	1	112.357	112.357	1.203	0.273
Seeding:Location:Site	1	53.694	53.694	0.575	0.449
Enhancement:Seeding:Location:Site	1	2.401	2.401	0.026	0.873
Residuals	354	33060.739	93.392		

Appendix: 1.12: Settled communities Simpsons diversity with habitat utilisation.

	Df	Sum Sq	Mean Sq	F value	Pr(>F)
Enhancement	1	0.116	0.116	2.284	0.372
Seeding	1	0.560	0.560	6.807	0.233
Location	1	1.568	1.568	149.285	0.052 †
Site	1	0.257	0.257	7.549	0.006*
Enhancement:Seeding	1	0.154	0.154	34.124	0.108
Enhancement:Location	1	0.000	0.000	0.003	0.966
Enhancement:Site	1	0.051	0.051	1.485	0.224
Seeding:Location	1	0.199	0.199	13.936	0.167
Seeding:Site	1	0.082	0.082	2.411	0.121
Location:Site	1	0.011	0.011	0.308	0.579
Enhancement:Seeding:Location	1	0.248	0.248	4.774	0.273
Enhancement:Seeding:Site	1	0.005	0.005	0.132	0.717
Enhancement:Location:Site	1	0.070	0.070	2.048	0.153
Seeding:Location:Site	1	0.014	0.014	0.419	0.518
Enhancement:Seeding:Location:Site	1	0.052	0.052	1.520	0.218
Residuals	354	12.075	0.034		

Appendix: 1.13: Settled communities Shannons diversity with habitat utilisation.

	Df	Sum Sq	Mean Sq	F value	Pr(>F)
Enhancement	1	0.549	0.549	2.314	0.370
Seeding	1	2.870	2.870	14.399	0.164
Location	1	11.433	11.433	198.813	0.045*
Site	1	0.457	0.457	3.601	0.059†
Enhancement:Seeding	1	0.726	0.726	5.629	0.254
Enhancement:Location	1	0.046	0.046	0.190	0.738
Enhancement:Site	1	0.237	0.237	1.869	0.172
Seeding:Location	1	0.540	0.540	22.555	0.132
Seeding:Site	1	0.199	0.199	1.570	0.211
Location:Site	1	0.058	0.058	0.453	0.501
Enhancement:Seeding:Location	1	1.108	1.108	2.433	0.363
Enhancement:Seeding:Site	1	0.129	0.129	1.016	0.314
Enhancement:Location:Site	1	0.242	0.242	1.909	0.168
Seeding:Location:Site	1	0.024	0.024	0.188	0.664
Enhancement:Seeding:Location:Site	1	0.455	0.455	3.588	0.059
Residuals	354	44.933	0.127		

Appendix: 1.14: Mobile invertebrate Shannons diversity with habitat utilisation.

	Df	Sum Sq	Mean Sq	F value	Pr(>F)
Enhancement	1	0.005	0.005	0.005	0.953
Seeding	1	2.406	2.406	51.985	0.088
Location	1	17.591	17.591	2368.291	0.013*
Site	1	0.018	0.018	0.364	0.654
Enhancement:Seeding	1	0.237	0.237	2.665	0.350
Enhancement:Location	1	0.051	0.051	6594.544	0.008*
Enhancement:Site	1	1.011	1.011	21.200	0.136
Seeding:Location	1	0.980	0.980	20.544	0.138
Seeding:Site	1	0.046	0.046	0.204	0.730
Location:Site	1	0.007	0.007	0.051	0.821
Enhancement:Seeding:Location	1	0.050	0.050	0.219	0.721
Enhancement:Seeding:Site	1	0.089	0.089	0.611	0.578
Enhancement:Location:Site	1	0.000	0.000	0.000	0.994
Seeding:Location:Site	1	0.048	0.048	0.328	0.567
Enhancement:Seeding:Location:Site	1	0.227	0.227	1.562	0.212
Residuals	354	51.522	0.146		

Appendix: 1.15: Settled communities richness with seawall

	Df	Sum Sq	Mean Sq	F value	Pr(>F)
Enhancement	3	200.618	66.873	20.811	0.016*
Site	1	20.864	20.864	8.034	0.005*
Month	1	45.669	45.669	321.757	0.035*
Enhancement:Site	3	9.640	3.213	1.237	0.298
Enhancement:Month	3	9.123	3.041	0.493	0.712
Site:Month	1	0.142	0.142	0.055	0.815
Enhancement:Site:Month	3	18.489	6.163	2.373	0.072
Residuals	174	451.897	2.597		

Appendix: 1.16: Mobile invertebrate richness with seawall

	Df	Sum Sq	Mean Sq	F value	Pr(>F)
Enhancement	3	120.235	40.078	16.885	0.022*
Site	1	1.934	1.934	2.123	0.147
Month	1	79.921	79.921	51.520	0.088
Enhancement:Site	3	7.121	2.374	2.606	0.053†
Enhancement:Month	3	7.425	2.475	1.231	0.434
Site:Month	1	1.551	1.551	1.703	0.194
Enhancement:Site:Month	3	6.032	2.011	2.208	0.089
Residuals	174	158.461	0.911		

Appendix: 1.17: Mobile invertebrate abundance with seawall

	Df	Sum Sq	Mean Sq	F value	Pr(>F)
Enhancement	3	6129.752	2043.251	146.082	0.001*
Site	1	430.350	430.350	10.536	0.001*
Month	1	4995.951	4995.951	9.609	0.199
Enhancement:Site	3	41.961	13.987	0.342	0.795
Enhancement:Month	3	1679.720	559.907	4.586	0.121
Site:Month	1	519.924	519.924	12.729	0.000*
Enhancement:Site:Month	3	366.283	122.094	2.989	0.033*
Residuals	174	7106.921	40.844		

Appendix: 1.18: Settled communities Simpsons diversity with seawall

	Df	Sum Sq	Mean Sq	F value	Pr(>F)
Enhancement	3	2.595	0.865	3.993	0.143
Site	1	0.042	0.042	1.470	0.227
Month	1	0.017	0.017	0.420	0.634
Enhancement:Site	3	0.650	0.217	7.580	0.000*
Enhancement:Month	3	0.149	0.050	0.850	0.551
Site:Month	1	0.040	0.040	1.404	0.238
Enhancement:Site:Month	3	0.175	0.058	2.043	0.110
Residuals	174	4.972	0.029		

Appendix: 1.19: Mobile invertebrate Simpsons diversity with seawall

	Df	Sum Sq	Mean Sq	F value	Pr(>F)
Enhancement	3	0.287	0.096	0.220	0.877
Site	1	0.174	0.174	1.672	0.198
Month	1	0.014	0.014	0.974	0.504
Enhancement:Site	3	1.307	0.436	4.178	0.007*
Enhancement:Month	3	1.905	0.635	17.414	0.021*
Site:Month	1	0.014	0.014	0.139	0.710
Enhancement:Site:Month	3	0.109	0.036	0.350	0.789
Residuals	174	18.138	0.104		

Appendix: 1.20: Settled communities Shannons diversity with seawall

	Df	Sum Sq	Mean Sq	F value	Pr(>F)
Enhancement	3	13.188	4.396	13.027	0.032*
Site	1	0.207	0.207	1.975	0.162
Month	1	0.126	0.126	1.247	0.465
Enhancement:Site	3	1.012	0.337	3.219	0.024*
Enhancement:Month	3	0.650	0.217	0.581	0.667
Site:Month	1	0.101	0.101	0.963	0.328
Enhancement:Site:Month	3	1.119	0.373	3.557	0.016*
Residuals	174	18.243	0.105		

Appendix: 1.21: Mobile invertebrate Shannons diversity with seawall

	Df	Sum Sq	Mean Sq	F value	Pr(>F)
Enhancement	3	9.871	3.290	19.045	0.019*
Site	1	0.235	0.235	2.294	0.132
Month	1	7.093	7.093	88.965	0.067*
Enhancement:Site	3	0.518	0.173	1.688	0.171
Enhancement:Month	3	0.929	0.310	1.054	0.483
Site:Month	1	0.080	0.080	0.779	0.379
Enhancement:Site:Month	3	0.881	0.294	2.871	0.038*
Residuals	174	17.807	0.102		

Appendix: 1.22: Fouling community Bray-Curtis test. `

Source	df	SS	MS	Pseudo-F	P(perm)	Unique perms	P(MC)
Enhancement	2	33948	16974	8.5289	0.094	180	0.0079*
Seeding	1	13823	13823	6.2081	0.4957	6	0.0482
Month	4	76521	19130	2.8058	0.0207	9778	0.0156*
Site	1	63693	63693	92.955	0.0001	9946	0.0001*
EnhancementxSeeding	2	12428	6214	4.9984	0.1044	9959	0.0292*
EnhancementxMonth	8	9027.8	1128.5	1.338	0.1799	9918	0.2038
EnhancementxSite	2	3980.4	1990.2	2.9045	0.0014	9917	0.0028*
SeedingxMonth	4	5735.7	1433.9	4.6574	0.0077	9955	0.004*
SeedingxSite	1	2226.7	2226.7	3.2496	0.0085	9958	0.0088*
MonthxSite	4	27272	6818.1	9.9504	0.0001	9899	0.0001*
EnhancementxSeedingxMonth	8	6374.6	796.83	1.0248	0.4887	9941	0.4748
EnhancementxSeedingxSite	2	2486.4	1243.2	1.8144	0.0652	9950	0.0604
EnhancementxMonthxSite	8	6747.2	843.41	1.2309	0.1756	9892	0.179
SeedingxMonthxSite	4	1231.5	307.88	0.44932	0.9631	9914	0.9616
EnhancementxSeedingxMonthxSite	8	6220.4	777.56	1.1348	0.281	9863	0.2794
Res	225	1.5417E+05	685.2				
Total	284	4.2758E+05					

Within level 'Seeded' of factor 'Seeding'

Groups	t	P(perm)	Unique perms	P(MC)
Flat, 2.5	1.899	0.2461	3	0.1168
Flat, 5	3.0341	0.3321	6	0.0294*
2.5, 5	1.4331	0.4935	6	0.2322

Within level 'Unseeded' of factor 'Seeding'

Groups	t	P(perm)	Unique perms	P(MC)
Flat, 2.5	2.4317	0.2485	3	0.0451*
Flat, 5	4.4044	0.3324	6	0.0144*
2.5, 5	2.1016	0.4977	6	0.1015

Appendix: 1.23: Mobile community Bray-Curtis test

Source	df	SS	MS	Pseudo-F	P(perm)	Unique perms	P(MC)
	2	50954	25477	9.1896	0.0911	180	0.0119*
Seeding	1	26822	26822	16.927	0.4905	6	0.0194*
Month	4	64867	16217	3.0982	0.0455	9768	0.0263*
Site	1	31552	31552	68.742	0.0001	9954	0.0001*
EnhancementxSeeding	2	14451	7225.4	20.753	0.0175	9954	0.0047*
EnhancementxMonth	8	6661.7	832.71	0.83161	0.6864	9919	0.6818
EnhancementxSite	2	5544.7	2772.4	6.0401	0.0001	9945	0.0001*
SeedingxMonth	4	3541	885.24	1.5558	0.2532	9954	0.2244
SeedingxSite	1	1584.6	1584.6	3.4523	0.0103	9952	0.0103*
MonthxSite	4	20937	5234.2	11.404	0.0001	9933	0.0001*
EnhancementxSeedingxMonth	8	4565.8	570.73	1.1294	0.4108	9949	0.3899
EnhancementxSeedingxSite	2	696.31	348.16	0.75852	0.6379	9950	0.6386
EnhancementxMonthxSite	8	8010.5	1001.3	2.1815	0.0003	9894	0.0004*
SeedingxMonthxSite	4	2275.9	568.98	1.2396	0.2572	9922	0.2558
EnhancementxSeedingxMonthxSite	8	4042.7	505.33	1.101	0.3488	9903	0.3437
Res	225	1.0327E+05	458.99				
Total	284	3.5516E+05					

Within level 'Seeded' of factor 'Seeding'

Groups	t	P(perm)	Unique perms	P(MC)
Flat, 2.5	2.0918	0.2444	3	0.0943
Flat, 5	1.5421	0.3282	6	0.2139
2.5, 5	1.1319	0.335	6	0.3876

Within level 'Unseeded' of factor 'Seeding'

Groups	t	P(perm)	Unique perms	P(MC)
Flat, 2.5	5.8436	0.252	3	0.0115*
Flat, 5	4.5801	0.3333	6	0.024*
2.5, 5	1.6518	0.497	6	0.1826

Appendix: 1.24: Fouling community Jaccards test

Source	df	SS	MS	Pseudo-F	P(perm)	Unique perms	P(MC)
Enhancement	2	27552	13776	5.116	0.1216	180	0.0124*
Seeding	1	11665	11665	4.537	0.5045	6	0.0618
Month	4	65096	16274	2.8505	0.0117	9761	0.0093*
Site	1	54944	54944	68.677	0.0001	9935	0.0001*
EnhancementxSeeding	2	10607	5303.4	70.56	0.0037	9965	0.0001*
EnhancementxMonth	8	12571	1571.4	1.4898	0.1092	9919	0.1008
EnhancementxSite	2	5385.4	2692.7	3.3658	0.0002	9935	0.0001*
SeedingxMonth	4	4371.4	1092.9	1.7769	0.1467	9932	0.0994
SeedingxSite	1	2571	2571	3.2136	0.0029	9938	0.0047
MonthxSite	4	22837	5709.2	7.1362	0.0001	9912	0.0001*
EnhancementxSeedingxMonth	8	7399.8	924.98	1.0147	0.4848	9912	0.4854
EnhancementxSeedingxSite	2	150.32	75.162	0.093948	0.9974	9944	0.9969
EnhancementxMonthxSite	8	8438.5	1054.8	1.3185	0.0809	9875	0.0888
SeedingxMonthxSite	4	2460.2	615.05	0.76878	0.7675	9913	0.7563
EnhancementxSeedingxMonthxSite	8	7293	911.62	1.1395	0.2503	9879	0.259
Res	225	1.8001E+05	800.03				
Total	284	4.2524E+05					

Within level 'Seeded' of factor 'Seeding'

Groups	t	P(perm)	Unique perms	P(MC)
Flat, 2.5	1.3428	0.2494	3	0.259
Flat, 5	2.0412	0.3354	6	0.0867
2.5, 5	1.1496	0.506	6	0.3805

Within level 'Unseeded' of factor 'Seeding'

Groups	t	P(perm)	Unique perms	P(MC)
Flat, 2.5	2.7932	0.2435	3	0.0285*
Flat, 5	4.8134	0.3333	6	0.007*
2.5, 5	1.6735	0.4927	6	0.1453

Appendix: 1.25: Mobile community Jaccards test

Source	df	SS	MS	Pseudo-F	P(perm)	Unique perms	P(MC)
Enhancement	2	42977	21488	6.8312	0.1501	180	0.0092*
Seeding	1	15675	15675	18.289	0.3388	6	0.0096*
Month	4	67664	16916	2.9894	0.0399	9785	0.0188*
Site	1	39957	39957	54.281	0.0001	9948	0.0001*
EnhancementxSeeding	2	16845	8422.7	11.042	0.0314	9971	0.0039*
EnhancementxMonth	8	11978	1497.2	1.0815	0.4106	9897	0.4063
EnhancementxSite	2	6291.2	3145.6	4.2733	0.0001	9939	0.0001*
SeedingxMonth	4	3659.2	914.8	1.2516	0.349	9954	0.3265
SeedingxSite	1	857.09	857.09	1.1644	0.3321	9933	0.3367
MonthxSite	4	22635	5658.7	7.6875	0.0001	9917	0.0001*
EnhancementxSeedingxMonth	8	6619.9	827.49	1.0149	0.4867	9923	0.4795
EnhancementxSeedingxSite	2	1525.6	762.78	1.0362	0.4168	9951	0.4186
EnhancementxMonthxSite	8	11075	1384.4	1.8807	0.0012	9895	0.0019*
SeedingxMonthxSite	4	2923.7	730.94	0.99298	0.4675	9918	0.477
EnhancementxSeedingxMonthxSite	8	6522.6	815.32	1.1076	0.3087	9903	0.3151
Res	225	1.6562E+05	736.1				
Total	284	4.2729E+05					

Within level 'Seeded' of factor 'Seeding'

Groups	t	P(perm)	Unique perms	P(MC)
Flat, 2.5	2.2263	0.2493	3	0.0642
Flat, 5	1.5062	0.3345	6	0.2091
2.5, 5	0.69375	0.6664	6	0.7565

Within level 'Unseeded' of factor 'Seeding'

Groups	t	P(perm)	Unique perms	P(MC)
Flat, 2.5	6.065	0.2511	3	0.0088*
Flat, 5	3.2911	0.4929	6	0.0441*
2.5, 5	0.90001	0.6682	6	0.5684

Appendix: 1.26: Fouling community Bray-Curtis test with seawall

Source	df	SS	MS	Pseudo-F	P(perm)	Unique perms	P(MC)
Enhancement	3	1.1013E+05	36711	8.5575	0.0115	4309	0.0045*
Month	4	28236	7059.1	1.2346	0.3308	7245	0.3312
Site	1	39459	39459	62.61	0.0001	9958	0.0001*
EnhancementxMonth	12	20150	1679.1	2.1256	0.0128	9911	0.0104*
EnhancementxSite	3	12870	4289.9	6.8069	0.0001	9937	0.0001*
MonthxSite	4	22872	5718	9.0727	0.0001	9926	0.0001*
EnhancementxMonthxSite	12	9479.7	789.98	1.2535	0.1436	9873	0.1469
Res	150	94536	630.24				
Total	189	3.386E+05					

Within level 'Devonport' of factor 'Site'

Groups	t	P(perm)	Unique perms	P(MC)
Flat, 2.5	1.5084	0.0855	31	0.1209
Flat, 5	3.4286	0.0079	56	0.0018*
Flat, Seawall	1.9756	0.0485	25	0.0405*
2.5, 5	1.3985	0.1295	91	0.1537
2.5, Seawall	0.9749	0.4727	46	0.4236
5, Seawall	3.2581	0.0001	9970	0.0001

Within level 'Westhaven' of factor 'Site'

Groups	t	P(perm)	Unique perms	P(MC)
Flat, 2.5	3.5134	0.0081	16	0.001*
Flat, 5	3.6232	0.0086	41	0.0013*
Flat, Seawall	2.461	0.0458	29	0.0222*
2.5, 5	0.2692	0.8755	23	0.8937
2.5, Seawall	2.1166	0.0135	41	0.0189*
5, Seawall	7.9981	0.0001	9959	0.0001*

Appendix: 1.27: Mobile community Bray-Curtis test with seawall

Source	df	SS	MS	Pseudo-F	P(perm)	Unique perms	P(MC)
Enhancement	3	61102	20367	12.366	0.0156	4342	0.0032*
Month	4	31891	7972.8	2.1107	0.1266	7267	0.1098
Site	1	19391	19391	40.321	0.0001	9956	0.0001*
EnhancementxMonth	12	14363	1197	1.205	0.3131	9918	0.2992
EnhancementxSite	3	4941.2	1647.1	3.4248	0.0004	9940	0.0008*
MonthxSite	4	15109	3777.3	7.8543	0.0001	9933	0.0001*
EnhancementxMonthxSite	12	11920	993.35	2.0655	0.0008	9893	0.0004*
Res	150	72139	480.92				
Total	189	2.3383E+05					

Within level 'Devonport' of factor 'Site' NS Month 12

Groups	t	P(perm)	Unique perms	P(MC)
Flat, 2.5	2.0251	0.0228	126	0.0271*
Flat, 5	4.1794	0.0086	126	0.0007*
Flat, Seawall	2.114	0.0277	91	0.0204*
2.5, 5	1.9431	0.0554	126	0.0477*
2.5, Seawall	1.0212	0.4064	126	0.3896
5, Seawall	1.7044	0.0541	126	0.0745

Within level 'Westhaven' of factor 'Site' NS Month 12

Groups	t	P(perm)	Unique perms	P(MC)
Flat, 2.5	5.59	0.0077	56	0.0001*
Flat, 5	5.5521	0.0082	56	0.0001*
Flat, Seawall	4.0429	0.009	56	0.0017*
2.5, 5	0.70111	0.704	126	0.6456
2.5, Seawall	2.4111	0.0086	126	0.0158*
5, Seawall	2.5701	0.022	126	0.0099*

Appendix: 1.28: Fouling community Jaccards test with seawall

Source	df	SS	MS	Pseudo-F	P(perm)	Unique perms	P(MC)
Enhancement	3	65834	21945	5.4811	0.0218	4322	0.0023*
Month	4	28778	7194.6	1.3022	0.2248	7295	0.2686
Site	1	36551	36551	46.332	0.0001	9939	0.0001*
EnhancementxMonth	12	22828	1902.4	1.4065	0.0734	9882	0.0896
EnhancementxSite	3	12011	4003.7	5.0751	0.0001	9910	0.0001*
MonthxSite	4	22101	5525.1	7.0037	0.0001	9908	0.0001*
EnhancementxMonthxSite	12	16230	1352.5	1.7144	0.0004	9849	0.0007*
Res	150	1.1833E+05	788.89				
Total	189	3.2377E+05					

Within level 'Devonport' of factor 'Site' NS Month 12

Groups	t	P(perm)	Unique perms	P(MC)
Flat, 2.5	1.6665	0.0371	126	0.0583 †
Flat, 5	1.7763	0.0345	126	0.054 †
Flat, Seawall	1.8925	0.0319	114	0.0309*
2.5, 5	0.72327	0.8145	126	0.6994
2.5, Seawall	1.2779	0.2074	126	0.2034
5, Seawall	1.4493	0.1095	126	0.1288

Within level 'Westhaven' of factor 'Site' NS Month 12

Groups	t	P(perm)	Unique perms	P(MC)
Flat, 2.5	2.5204	0.0084	66	0.0059*
Flat, 5	3.7746	0.0077	30	0.0013*
Flat, Seawall	2.7819	0.0071	66	0.0048*
2.5, 5	1.4672	0.1689	29	0.1523
2.5, Seawall	2.308	0.0085	126	0.0159*
5, Seawall	2.4051	0.0164	52	0.0261*

Appendix: 1.29: Mobile community Jaccards test with seawall

Source	df	SS	MS	Pseudo-F	P(perm)	Unique perms	P(MC)
Enhancement	3	56729	18910	7.1434	0.0462	4316	0.0053*
Month	4	35657	8914.2	2.0985	0.1044	7271	0.0885
Site	1	21477	21477	30.018	0.0001	9956	0.0001*
EnhancementxMonth	12	17486	1457.2	1.0248	0.471	9912	0.4674
EnhancementxSite	3	7941.4	2647.1	3.6998	0.0001	9938	0.0002*
MonthxSite	4	16991	4247.8	5.937	0.0001	9923	0.0001*
EnhancementxMonthxSite	12	17062	1421.9	1.9873	0.0006	9877	0.0007*
Res	150	1.0732E+05	715.48				
Total	189	2.8507E+05					

12 months

Within level 'Devonport' of factor 'Site' NS Month 12

Groups	t	P(perm)	Unique perms	P(MC)
Flat, 2.5	1.5084	0.0855	31	0.1209
Flat, 5	3.4286	0.0079	56	0.0018*
Flat, Seawall	1.9756	0.0485	25	0.0405*
2.5, 5	1.3985	0.1295	91	0.1537
2.5, Seawall	0.9749	0.4727	46	0.4236
5, Seawall	1.3491	0.1622	56	0.1867

Within level 'Westhaven' of factor 'Site' NS Month 12

Groups	t	P(perm)	Unique perms	P(MC)
Flat, 2.5	3.5134	0.0081	16	0.001*
Flat, 5	3.6232	0.0086	41	0.0013*
Flat, Seawall	2.461	0.0458	29	0.0222*
2.5, 5	0.2692	0.8755	23	0.8937
2.5, Seawall	2.1166	0.0135	41	0.0189*
5, Seawall	2.6029	0.0083	66	0.0076*

Appendix: 1.1: Species richness of the fouling communities.

	Df	Sum Sq	Mean Sq	F value	Pr(>F)
Enhancement	2	60.089	30.044	33.383	0.000*
Rain	1	0.178	0.178	0.198	0.658
Month	2	12.156	6.078	6.753	0.002*
Enhancement:Rain	2	8.622	4.311	4.790	0.011*
Enhancement:Month	4	7.778	1.944	2.160	0.082
Rain:Month	2	0.156	0.078	0.086	0.917
Enhancement:Rain:Month	4	0.444	0.111	0.123	0.974
Residuals	72	64.800	0.900		

Appendix: 1.2: Species richness of the mobile invertebrates.

	Df	Sum Sq	Mean Sq	F value	Pr(>F)
Enhancement	2	54.067	27.033	26.884	0.000*
Rain	1	0.044	0.044	0.044	0.834
Month	2	19.467	9.733	9.680	0.000*
Enhancement:Rain	2	0.422	0.211	0.210	0.811
Enhancement:Month	4	3.067	0.767	0.762	0.553

Rain:Month	2	1.156	0.578	0.575	0.565
Enhancement:Rain:Month	4	3.778	0.944	0.939	0.446
Residuals	72	72.400	1.006		

Appendix: 2.3: Fouling coverage (%) across the settlement plates.

	Df	Sum Sq	Mean Sq	F value	Pr(>F)
Enhancement	2	5526.600	2763.300	28.473	0.000*
Rain	1	127.211	127.211	1.311	0.256
Month	2	510.067	255.033	2.628	0.079
Enhancement:Rain	2	343.089	171.544	1.768	0.178
Enhancement:Month	4	2687.333	671.833	6.923	0.000*
Rain:Month	2	55.756	27.878	0.287	0.751
Enhancement:Rain:Month	4	332.444	83.111	0.856	0.494
Residuals	72	6987.600	97.050		

Appendix: 2.4: Mobile invertebrate abundance across the settlement plates.

	Df	Sum Sq	Mean Sq	F value	Pr(>F)
Enhancement	2	298.022	149.011	17.462	0.000*
Rain	1	6.944	6.944	0.814	0.370
Month	2	116.822	58.411	6.845	0.002*
Enhancement:Rain	2	8.822	4.411	0.517	0.599
Enhancement:Month	4	53.644	13.411	1.572	0.191
Rain:Month	2	4.289	2.144	0.251	0.778
Enhancement:Rain:Month	4	28.844	7.211	0.845	0.501
Residuals	72	614.400	8.533		

Appendix: 2.5: Fouling communities Simpsons diversity.

	Df	Sum Sq	Mean Sq	F value	Pr(>F)
Enhancement	2	1.739	0.869	6.435	0.003*
Rain	1	0.030	0.030	0.225	0.637
Month	2	0.330	0.165	1.220	0.301
Enhancement:Rain	2	0.440	0.220	1.627	0.204
Enhancement:Month	4	0.917	0.229	1.697	0.160
Rain:Month	2	0.061	0.030	0.225	0.799
Enhancement:Rain:Month	4	0.449	0.112	0.830	0.510
Residuals	72	9.728	0.135		

Appendix: 2.6: Mobile invertebrates Simpsons diversity.

	Df	Sum Sq	Mean Sq	F value	Pr(>F)
Enhancement	2	1.572	0.786	6.798	0.002*
Rain	1	0.071	0.071	0.613	0.436
Month	2	0.395	0.197	1.707	0.189
Enhancement:Rain	2	0.552	0.276	2.385	0.099
Enhancement:Month	4	0.484	0.121	1.046	0.389
Rain:Month	2	0.583	0.292	2.521	0.087
Enhancement:Rain:Month	4	0.585	0.146	1.266	0.291
Residuals	72	8.327	0.116		

Appendix: 2.7: Fouling communities Shannons diversity.

	Df	Sum Sq	Mean Sq	F value	Pr(>F)
Enhancement	2	3.529	1.764	19.816	0.000*
Rain	1	0.182	0.182	2.049	0.157
Month	2	2.330	1.165	13.087	0.000*
Enhancement:Rain	2	0.662	0.331	3.720	0.029*
Enhancement:Month	4	1.511	0.378	4.243	0.004*
Rain:Month	2	0.054	0.027	0.306	0.737
Enhancement:Rain:Month	4	0.103	0.026	0.290	0.883
Residuals	72	6.411	0.089		

Appendix: 2.8: Mobile invertebrates Shannons diversity.

	Df	Sum Sq	Mean Sq	F value	Pr(>F)
Enhancement	2	4.798	2.399	18.502	0.000*
Rain	1	0.003	0.003	0.026	0.873
Month	2	1.912	0.956	7.374	0.001*
Enhancement:Rain	2	0.220	0.110	0.848	0.433
Enhancement:Month	4	0.474	0.119	0.914	0.461
Rain:Month	2	0.485	0.242	1.869	0.162
Enhancement:Rain:Month	4	0.172	0.043	0.331	0.856
Residuals	72	9.335	0.130		

Appendix: 2.9: Fouling communities richness with habitat utilisation.

	Df	Sum Sq	Mean Sq	F value	Pr(>F)
Enhancement	1	16.133	16.133	22.068	0.000*
Habitat	1	38.533	38.533	52.707	0.000*
Rain	1	3.333	3.333	4.559	0.035*
Month	1	3.613	3.613	4.941	0.028*
Enhancement:Habitat	1	6.533	6.533	8.936	0.003*
Enhancement:Rain	1	2.133	2.133	2.918	0.091
Enhancement:Month	1	1.013	1.013	1.385	0.242
Habitat:Rain	1	2.133	2.133	2.918	0.091
Habitat:Month	1	2.113	2.113	2.890	0.092
Rain:Month	1	0.113	0.113	0.154	0.696
Enhancement:Habitat:Rain	1	6.533	6.533	8.936	0.003*
Enhancement:Habitat:Month	1	0.612	0.612	0.838	0.362
Enhancement:Rain:Month	1	0.313	0.313	0.427	0.515
Habitat:Rain:Month	1	0.113	0.113	0.154	0.696
Enhancement:Habitat:Rain:Month	1	0.612	0.612	0.838	0.362
Residuals	104	76.033	0.731		

Appendix: 2.10: Mobile invertebrate richness with habitat utilisation

	Df	Sum Sq	Mean Sq	F value	Pr(>F)
Enhancement	1	8.008	8.008	8.144	0.005*
Habitat	1	165.675	165.675	168.483	0.000*
Rain	1	0.075	0.075	0.076	0.783
Month	1	24.200	24.200	24.610	0.000*
Enhancement:Habitat	1	0.208	0.208	0.212	0.646
Enhancement:Rain	1	0.675	0.675	0.686	0.409
Enhancement:Month	1	0.000	0.000	0.000	1.000
Habitat:Rain	1	0.008	0.008	0.008	0.927
Habitat:Month	1	1.800	1.800	1.831	0.179
Rain:Month	1	0.200	0.200	0.203	0.653
Enhancement:Habitat:Rain	1	0.075	0.075	0.076	0.783
Enhancement:Habitat:Month	1	3.200	3.200	3.254	0.074
Enhancement:Rain:Month	1	1.800	1.800	1.831	0.179
Habitat:Rain:Month	1	0.000	0.000	0.000	1.000
Enhancement:Habitat:Rain:Month	1	0.800	0.800	0.814	0.369
Residuals	104	102.267	0.983		

Appendix: 2.11: Fouling communities coverage with habitat utilisation.

	Df	Sum Sq	Mean Sq	F value	Pr(>F)
Enhancement	1	1326.675	1326.675	31.265	0.000*
Habitat	1	735.075	735.075	17.323	0.000*
Rain	1	147.408	147.408	3.474	0.065
Month	1	234.612	234.612	5.529	0.021*
Enhancement:Habitat	1	516.675	516.675	12.176	0.001*
Enhancement:Rain	1	88.408	88.408	2.083	0.152
Enhancement:Month	1	1029.613	1029.613	24.264	0.000*
Habitat:Rain	1	66.008	66.008	1.556	0.215
Habitat:Month	1	340.312	340.312	8.020	0.006*
Rain:Month	1	2.112	2.112	0.050	0.824
Enhancement:Habitat:Rain	1	147.408	147.408	3.474	0.065†
Enhancement:Habitat:Month	1	201.613	201.613	4.751	0.032*
Enhancement:Rain:Month	1	94.612	94.612	2.230	0.138
Habitat:Rain:Month	1	0.113	0.113	0.003	0.959
Enhancement:Habitat:Rain:Month	1	25.313	25.313	0.597	0.442
Residuals	104	4413.033	42.433		

Appendix: 2.12: Mobile invertebrate abundance with habitat utilisation.

	Df	Sum Sq	Mean Sq	F value	Pr(>F)
Enhancement	1	112.133	112.133	11.028	0.001*
Habitat	1	2167.500	2167.500	213.177	0.000*
Rain	1	0.833	0.833	0.082	0.775
Month	1	678.612	678.612	66.742	0.000*
Enhancement:Habitat	1	58.800	58.800	5.783	0.018*
Enhancement:Rain	1	0.533	0.533	0.052	0.819
Enhancement:Month	1	7.813	7.813	0.768	0.383
Habitat:Rain	1	0.033	0.033	0.003	0.954
Habitat:Month	1	475.313	475.313	46.748	0.000*
Rain:Month	1	0.613	0.613	0.060	0.807
Enhancement:Habitat:Rain	1	0.133	0.133	0.013	0.909
Enhancement:Habitat:Month	1	0.012	0.012	0.001	0.972
Enhancement:Rain:Month	1	15.312	15.312	1.506	0.223
Habitat:Rain:Month	1	0.312	0.312	0.031	0.861
Enhancement:Habitat:Rain:Month	1	17.113	17.113	1.683	0.197
Residuals	104	1057.433	10.168		

Appendix: 2.13: Fouling communities Simpsons diversity with habitat utilisation.

	Df	Sum Sq	Mean Sq	F value	Pr(>F)
Enhancement	1	0.844	0.844	5.958	0.016*
Habitat	1	0.024	0.024	0.167	0.684
Rain	1	0.259	0.259	1.829	0.179
Month	1	0.605	0.605	4.272	0.041*
Enhancement:Habitat	1	0.094	0.094	0.661	0.418
Enhancement:Rain	1	0.004	0.004	0.028	0.868
Enhancement:Month	1	1.380	1.380	9.735	0.002*
Habitat:Rain	1	0.003	0.003	0.022	0.881
Habitat:Month	1	0.010	0.010	0.069	0.793
Rain:Month	1	0.113	0.113	0.795	0.375
Enhancement:Habitat:Rain	1	0.814	0.814	5.743	0.018*
Enhancement:Habitat:Month	1	0.463	0.463	3.268	0.074
Enhancement:Rain:Month	1	0.032	0.032	0.224	0.637
Habitat:Rain:Month	1	0.073	0.073	0.512	0.476
Enhancement:Habitat:Rain:Month	1	0.001	0.001	0.010	0.922
Residuals	104	14.738	0.142		

Appendix: 2.14: Mobile invertebrate Simpsons diversity with habitat utilisation.

	Df	Sum Sq	Mean Sq	F value	Pr(>F)
Enhancement	1	0.001	0.001	0.005	0.945
Habitat	1	2.108	2.108	19.399	0.000*
Rain	1	0.053	0.053	0.488	0.487
Month	1	1.068	1.068	9.829	0.002*
Enhancement:Habitat	1	0.320	0.320	2.943	0.089
Enhancement:Rain	1	0.002	0.002	0.017	0.897
Enhancement:Month	1	0.038	0.038	0.350	0.555
Habitat:Rain	1	0.056	0.056	0.514	0.475
Habitat:Month	1	0.481	0.481	4.425	0.038*
Rain:Month	1	0.247	0.247	2.271	0.135
Enhancement:Habitat:Rain	1	0.094	0.094	0.865	0.355
Enhancement:Habitat:Month	1	0.066	0.066	0.608	0.437
Enhancement:Rain:Month	1	0.020	0.020	0.188	0.666
Habitat:Rain:Month	1	0.004	0.004	0.035	0.851
Enhancement:Habitat:Rain:Month	1	0.085	0.085	0.785	0.378
Residuals	104	11.299	0.109		

Appendix: 2.15: Fouling communities Shannons diversity with habitat utilisation.

	Df	Sum Sq	Mean Sq	F value	Pr(>F)
Enhancement	1	0.715	0.715	8.565	0.004*
Habitat	1	4.087	4.087	48.995	0.000*
Rain	1	0.538	0.538	6.446	0.013*
Month	1	1.193	1.193	14.302	0.000*
Enhancement:Habitat	1	0.564	0.564	6.767	0.011*
Enhancement:Rain	1	0.122	0.122	1.462	0.229
Enhancement:Month	1	0.047	0.047	0.567	0.453
Habitat:Rain	1	0.575	0.575	6.896	0.010*
Habitat:Month	1	0.481	0.481	5.769	0.018*
Rain:Month	1	0.120	0.120	1.435	0.234
Enhancement:Habitat:Rain	1	0.224	0.224	2.690	0.104
Enhancement:Habitat:Month	1	0.016	0.016	0.186	0.667
Enhancement:Rain:Month	1	0.045	0.045	0.542	0.463
Habitat:Rain:Month	1	0.064	0.064	0.769	0.383
Enhancement:Habitat:Rain:Month	1	0.035	0.035	0.414	0.521
Residuals	104	8.676	0.083		

Appendix: 2.16: Mobile invertebrate Shannons diversity with habitat utilisation.

	Df	Sum Sq	Mean Sq	F value	Pr(>F)
Enhancement	1	1.029	1.029	9.199	0.003*
Habitat	1	13.782	13.782	123.162	0.000*
Rain	1	0.006	0.006	0.053	0.819
Month	1	0.821	0.821	7.340	0.008*
Enhancement:Habitat	1	0.062	0.062	0.557	0.457
Enhancement:Rain	1	0.070	0.070	0.628	0.430
Enhancement:Month	1	0.014	0.014	0.126	0.723
Habitat:Rain	1	0.048	0.048	0.430	0.513
Habitat:Month	1	0.016	0.016	0.139	0.710
Rain:Month	1	0.055	0.055	0.488	0.486
Enhancement:Habitat:Rain	1	0.015	0.015	0.135	0.714
Enhancement:Habitat:Month	1	0.811	0.811	7.246	0.008*
Enhancement:Rain:Month	1	0.113	0.113	1.007	0.318
Habitat:Rain:Month	1	0.064	0.064	0.573	0.451
Enhancement:Habitat:Rain:Month	1	0.100	0.100	0.893	0.347
Residuals	104	11.638	0.112		

Appendix: 3.1: Literature cited within the Waitematā taxonomy list.

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Appendix: 3.2: Waitematā taxonomy list.

Species	Endemism	Distribution	Habitat	Source
Phylum FORAMINIFERA				
Class GLOBOTHALAMEA				
ELPHIDIIDAE				
Cribroelphidium excavatum	Native	Temperate Cosmopolitan, China, South Korea, Japan		Hayward et al. (1997a)
Elphidium advenum	Native	European waters, Mediterranean, Gulf of Mexico, China, Japan, N. Atlantic, South Korea, New Zealand, Guam		Hayward et al. (1997a)
Elphidium williamsoni	Native	Canada, European waters, New Zealand		Hayward et al. (1997a)
Elphidium gunteri	Native	New Zealand, Australia, South Korea, Gulf of Mexico, Europe		Hayward et al. (1997a)
TROCHAMMINIDAE				
Entzia macrescens	Native	European waters, Mediterranean, Gulf of Mexico, China, Japan, N. Atlantic, South Korea, New Zealand, U.S., Malaysia,		Hayward et al. (1997a)
Trochammina inflata	Native	New Zealand, N. Atlantic, U.S., Gulf of Mexico, Canada, China, European waters, Mediterranean, South Korea, Japan, Malaysia		Hayward et al. (1997a)
Paratrochammina bartrami	Endemic	New Zealand		Hayward et al. (1997a)
Lepidodeuterammina ochracea	Native	New Zealand, Canada, European waters, Gulf of Mexico, U.S., China, N. Atlantic		Hayward et al. (1997a)
Portatrochammina sorosa	Endemic	New Zealand		Hayward et al. (1997a)
HELENINIDAE				
Helenina anderseni	Native	New Zealand, China, Japan, Gulf of Mexico, U.S., Spain		Hayward et al. (1997a)
Species	Endemism	Distribution	Habitat	Source

HAPLOPHRAGMOIDIDAE				
Haplophragmoides wilberti	Native	European waters, U.S., Gulf of Mexico, Malaysia, New Zealand	Brackish	Hayward et al. (1997a)
Trochamminita salsa	Native	Gulf of Mexico, New Zealand	Low salinity	Hayward et al. (1997a)
ROTALIIDAE				
Ammonia beccarii	Native	Adriatic Sea, China, New Zealand		Hayward et al. (1997a)
ROSALINIDAE				
Gavelinopsis praegeri	Native	China, Japan, Guam, New Zealand, South Korea, European waters, Mediterranean		Hayward et al. (1997a)
Rosalina irregularis	Native	New Zealand, South Korea, Europe		Hayward et al. (1997a)
POLYMORPHINIDAE				
Guttulina communis	Native	China, Japan, New Zealand, South Korea, European waters, Mediterranean, Gulf of Mexico, South East U.S.	subtidal sediment	Hayward et al. (1997a)
SIPHOGENERINOIDIDAE				
Siphogenerina striata	Native	Japan, New Zealand, South Korea		Hayward et al. (1997a)
Siphogenerina raphana	Non-indigenous	Cosmopolitan	Benthic soft sediment	Hayward, et al. (1997)
LITUOLIDAE				
Ammobaculites exiguus	Native	New Zealand, New Caledonia, Gulf of Mexico, U.S., South Korea, Malaysia, Japan, China		Hayward et al. (1997a)
Ammotium fragile	Native	New Zealand, New Caledonia, Europe		Hayward et al. (1997a)
Species	Endemism	Distribution	Habitat	Source

NOTOROTALIIDAE				
Notorotalia finlayi	Endemic	New Zealand		Hayward et al. (1997a)
HORMOSINIDAE				
Cuneata arctica	Native	Canada, China, Norway, U.S., N. Atlantic, New Zealand, Japan, Irish Sea		Hayward et al. (1997a)
KUNKLERINIDAE				
Scherochorella moniliformis	Native	New Zealand, U.K.		Hayward et al. (1997a)
TEXTULARIIDAE				
Textularia earlandi	Native	Canada, European waters, U.S., South Korea, Japan, China		Hayward et al. (1997a)
NONIONIDAE				
Astrononion novozealandicum	Native	New Caledonia, Japan, China, New Zealand		Hayward et al. (1997a)
Haynesina depressula	Native	New Caledonia, Japan, China, New Zealand, European waters, Mediterranean, South Korean, SW Pacific		Hayward et al. (1997a)
Nonionellina flemingi	Endemic	New Zealand		Hayward et al. (1997a)
BOLIVINITIDAE				
Bolivina arta	Endemic	New Zealand		Hayward et al. (1997a)
Bolivina cacozela	Endemic	New Zealand		Hayward et al. (1997a)
Bolivina compacta	Native	China, Japan, Guam, New Zealand, South Korea		Hayward et al. (1997a)
Bolivina spathulata	Native	China, Japan, New Caledonia, New Zealand, South Korea, Norway, Carolina		Hayward et al. (1997a)
Bolivina striatula	Native	Bay of Biscay, East Asia, New Caledonia, Florida, Guam, N. Atlantic		Hayward et al. (1997a)
Species	Endemism	Distribution	Habitat	Source

Bolivina variabilis	Unknown			Hayward et al. (1997a)
Bolivinellina translucens	Native	New Caledonia, Gulf of Mexico, New Zealand		Hayward et al. (1997a)
Fursenkoina schreibersiana	Native	European waters, East Asia, New Caledonia, East U.S. Continental Shelf, New Zealand		Hayward et al. (1997a)
TRICHOHYALIDAE				
Buccella frigida	Native	Canada, East Asia, N. Atlantic, U.S., New Zealand		Hayward et al. (1997a)
BULIMINIDAE				
Bulimina elongata	Native	New Zealand, Bay of Biscay, Norway, Japan, South Korea, Marmara Sea, U.K.		Hayward et al. (1997a)
Bulimina gibba	Native	New Zealand, Bay of Biscay, Norway, South Korea, U.K., Southeast U.S. Continental Shelf, Yellow Sea		Hayward et al. (1997a)
Bulimina marginata	Native	New Zealand, Bay of Biscay, East Asia, N. Atlantic, E. U.S., Marmara Sea, U.K.		Hayward et al. (1997a)
Buliminella elegantissima	Native	New Zealand, Bay of Biscay, East Asia, N. Atlantic, U.S., U.K., Norway, New Caledonia, Canada		Hayward et al. (1997a)
CASSIDULINIDAE				
Cassidulina carinata	Native	New Zealand, East Asia, Southeast U.S. Continental Shelf, England, New Caledonia		Hayward et al. (1997a)
Globocassidulina minuta	Native	New Zealand, Bay of Biscay, Guam, Marmara Sea, Gulf of Cadiz		Hayward et al. (1997a)
CIBICIDIDAE				
Lobatula lobatula	Native	New Zealand, Bay of Biscay, East Asia, N. Atlantic, Eastern U.S., U.K., Norway, New Caledonia, Canada, Guam		Hayward et al. (1997a)
Species	Endemism	Distribution	Habitat	Source

Cibicidoides dispars	Native	New Zealand, Bay of Biscay		Hayward et al. (1997a)
Cibicides vortex	Endemic	New Zealand		Hayward et al. (1997a)
PLACENTULINIDAE				
Patellinella inconspicua	Native	New Zealand, Japan, South Korea, Guam, New Caledonia		Hayward et al. (1997a)
SPHAEROIDINIDAE				
Sphaeroidina bulloides	Native	New Zealand, Bay of Biscay, East Asia, NE U.S. Continental Shelf, North Carolina, U.K., New Caledonia, Guam, Marmara Sea		Hayward et al. (1997a)
GLOBOROTALIOIDEA				
Globorotalia truncatulinoides	Native	New Zealand, China		Hayward et al. (1997a)
Class FORAMINIFERA INCERTAE SEDIS				
NODOSARIIDAE				
Laevidentalina filiformis	Native	New Zealand, East Asia, Eastern U.S. Continental Shelf, New Caledonia		Hayward et al. (1997a)
LAGENIDAE				
Lagena paucistriata	Native	New Zealand, Australia, New Caledonia		Hayward et al. (1997a)
VAGINULINIDAE				
Lenticulina gibba	Native	New Zealand, Bay of Biscay, U.K., Japan, South Korea, Marmara Sea, New Caledonia, Florida, Eastern U.S. Continental Shelf		Hayward et al. (1997a)
Species	Endemism	Distribution	Habitat	Source

ELLIPSOLAGENIDAE				
Favulina melo	Native	New Zealand, Bay of Biscay, U.K., East Asia, Marmara Sea, New Caledonia, Marmara Sea, E. U.S. Continental Shelf, North Carolina		Hayward et al. (1997a)
Class TUBOTHALAMEA				
MILIAMMINIDAE				
Miliammina fusca	Native	New Zealand, Bay of Biscay, U.K., Japan, South Korea, Norway, East U.S., North Carolina, N. Atlantic, Malaysia	Brackish	Hayward et al. (1997a)
Miliammina obliqua	Native	New Zealand, New Caledonia, Malaysia		Hayward et al. (1997a)
CORNUSPIRIDAE				
Cornuspira involvens	Native	New Zealand, Bay of Biscay, U.K., East Asia, New Caledonia, Norway, East U.S., Canada, N. Atlantic, Marmara Sea		Hayward et al. (1997a)
OPHTHALMIDIIDAE				
Edentostomina cultrata	Native	New Zealand, Japan, China, Bay of Biscay, North Carolina, New Caledonia		Hayward et al. (1997a)
HAUERINIDAE				
Miliolinella subrotunda	Native	New Zealand, Bay of Biscay, U.K., East Asia, New Caledonia, Norway, East U.S., Canada, N. Atlantic, Marmara Sea		Hayward et al. (1997a)
Quinqueloculina ariminensis	Native	New Zealand, Adriatic Sea		Hayward et al. (1997a)
Lachlanella cooki	Endemic	New Zealand		Hayward et al. (1997a)
Quinqueloculina patagonica	Native	New Zealand, Zanzibar Archipelago, King George Island		Hayward et al. (1997a)
Species	Endemism	Distribution	Habitat	Source

Quinqueloculina seminula	Native	New Zealand, Bay of Biscay, U.K., East Asia, New Caledonia, Norway, East U.S., Canada, N. Atlantic, Marmara Sea		Hayward et al. (1997a)
Triloculina trigonula	Native	New Zealand, Bay of Biscay, U.K., East Asia, New Caledonia, East U.S., Canada, N. Atlantic, Marmara Sea, Zanzibar Archipelago, Guam		Hayward et al. (1997a)
Class MONOTHALAMEA				
SACCAMMINIDAE				
Pseudothurammina limnetis	Native	New Zealand, New Caledonia, Japan		Hayward et al. (1997a)
Phlyum MYZOZOA				
Class DINOPHYCEAE				
GYMNODINIACEAE				
Cochlodinium sp.	Unknown			Inglis et al. (2006)
Gymnodinium catenatum	Cryptogenic	New Zealand, Australia, China, Japan, Singapore, Turkey, Argentina, Gulf of California, Mediterranean Sea, Mexico		Inglis et al. (2006)
GONYAULACALES				
Lingulodinium polyedrum	Native	New Zealand, Australia, Central America, European waters, China, Taiwan, South-West Asia, Brazil, Colombia, Mauritania		Inglis et al. (2006)
Protoperidinium conicum	Native	New Zealand, Australia, South-West Asia, Egypt, Atlantic Islands, South Asia Sea, China, Russia, Yellow Sea, Chile, Colombia, Mexico, Europe		Inglis et al. (2006)
Protoperidinium conicum conicoides	Unknown			Inglis et al. (2006)
Scrippsiella trochoidea	Native	New Zealand, Australia, Turkey, N. America, Atlantic Islands, China, Japan, Eastern Russia, Chile, Colombia, Mexico, European waters, Brazil, Mauritania, Namibia		Inglis et al. (2006)
Species	Endemism	Distribution	Habitat	Source

Phylum BIGYRA				
Class LABYRINTHULEA				
LABYRINTHULACEAE				
Labyrinthula sp.	Endemic	New Zealand		Armiger, L. C. (1964)
Phylum OCHROPHYTA				
Class PHAEOPHYCEAE				
SARGASSACEAE				
Carpophyllum flexuosum	Native	New Zealand, Tasmania	Subtidal rocky reef	Dromgoole & Foster (1983), Inglis et al. (2006)
Cutleria multifida	Non-indigenous	Cosmopolitan in temperate seas		Adams, N. M. (1983)
Carpophyllum maschalocarpum	Endemic	New Zealand	Subtidal rocky reef	Shears, N. T. (2012)
Sargassum sinclairii	Endemic	New Zealand	Subtidal rocky reef	Shears, N. T. (2012)
Carpophyllum plumosum	Endemic	New Zealand	Subtidal rocky reef	Shears, N. T. (2012)
SCYTOSIPHONACEAE				
Hydroclathrus clathratus	Non-indigenous	Cosmopolitan		Johnson & Dromgoole (1977)
Colpomenia bullosa	Cryptogenic	New Zealand, Canada, U.S., Pacific		Dromgoole & Foster (1983)
Colpomenia durvillei	Non-indigenous	N. Pacific, Japan, California, New Zealand		Halliday et al. (2006)
Colpomenia sinuosa	Native	Gulf of Mexico, Madagascar, Somalia, Indian Ocean, Japan, Kenya, Mediterranean, Mozambique, New Zealand, Australia, N Atlantic, South Africa		Shears, N. T. (2012)
Species	Endemism	Distribution	Habitat	Source

ALARIACEAE				
Undaria pinnatifida	Non-indigenous	Japan, Korea, parts of China		Halliday et al. (2006)
HORMOSIRACEAE				
Hormosira banksii	Native	New Zealand, Australia		Chapman & Ronaldson (1958)
LESSONIACEAE				
Ecklonia radiata	Native	New Zealand, Australia, Madagascar, South Africa, Canary Islands, Cape Verde Islands	Subtidal rocky reef	Shears, N. T. (2012)
RALFSIALES				
Ralfsia sp.	Unknown			Shears, N. T. (2012)
STYPOCAULACEAE				
Halopteris funicularis	Native	New Zealand, South Africa		Shears, N. T. (2012)
SCYTOTHAMNACEAE				
Scytothamnus australis	Endemic	New Zealand		Shears, N. T. (2012)
Phylum CHOLOROPHYTA				
Class ULVOPHYCEAE				
CODIACEAE				
Codium fragile subsp. fragile	Non-indigenous	NW Pacific, Africa, N. Atlantic, N. / S. America, Mediterranean, E. Pacific, Australia, New Zealand	Cosmopolitan hard substrate	Dromgoole, F. L. (1975)
Species	Endemism	Distribution	Habitat	Source

ULVACEAE				
Ulva intestinalis	Native	Cosmopolitan		Chapman & Ronaldson (1958)
Ulva flexuosa	Native	Cosmopolitan		Chapman & Ronaldson (1958)
GOMONTIACEAE				
Urospora penicilliformis	Native	Cosmopolitan		Chapman & Ronaldson (1958)
Monostroma latissimum	Native	Brazil, West European waters, Sweden, Japan, Korea, Taiwan, India, Philippines, New Zealand, S. Australia		Chapman & Ronaldson (1958)
CLADOPHORACEAE				
Chaetomorpha ligustica	Native	Cosmopolitan		Chapman & Ronaldson (1958)
Rhizoclonium africanum	Native	Atlantic islands, Texas, Central America, Caribbean Islands, Africa, W. Atlantic, Indian Ocean, S. China Sea, S. / SE Asia, New Zealand, Australia, Pacific Islands		Chapman & Ronaldson (1958)
Cladophora herpestica	Native	Europe, Mediterranean, Madagascar, Japan, Indian Ocean, Somalia, South Africa, Tanzania, New Zealand		Shears, N. T. (2012)
Phylum RHODOPHYTA				
Class FLORIDEOPHYCEAE				
CORALLINACEAE				
Corallina officinalis	Native	N. Atlantic, Japan, China, Australasia	Rocky reef, Sub-intertidal	Dromgoole & Foster (1983)
Species	Endemism	Distribution	Habitat	Source

SOLIERIACEAE				
Solieriaceae sp.	Non-indigenous			Nelson, W. A. (1994)
GRACILARIACEAE				
Gracilaria secundata	Native	Kiama, New South Wales, Australia, New Zealand		Chapman & Ronaldson (1958)
CAULACANTHACEAE				
Catenella nipae	Native	South Africa, Tanzania, Bangladesh, India, Asia, Australasia		Chapman & Ronaldson (1958)
Catenella fusiformis	Native	Temperate South America, Australia, New Zealand, Antarctica, Subantarctic Islands		Chapman & Ronaldson (1958)
RHODOMELACEAE				
Bostrychia harveyi	Native	South Africa, New Zealand, Australia		Chapman & Ronaldson (1958)
Polysiphonia implexa	Native	New Zealand, Australia		Chapman & Ronaldson (1958)
DELESSERIACEAE				
Caloglossa leprieurii	Native	Cosmopolitan		Chapman & Ronaldson (1958)
GELIDIACEAE				
Gelidium caulacantheum	Native	New Zealand, Australia		Chapman & Ronaldson (1958)
HALYMENIACEAE				
Pachymenia lusoria	Native	New Zealand, Australia, Subantarctic		
Species	Endemism	Distribution	Habitat	Source

PTEROCLADIACEAE				
Pterocladiella capillacea	Native	Cosmopolitan		
Phylum TRACHEOPHYTA				
RESTIONACEAE				
Leptocarpus similis	Endemic	New Zealand		Hayward et al. (1997b)
CYPERACEAE				
Bolboschoenus caldwellii	Native	New Zealand, Australia	Saltmarshes	V.J. Cook. (1947)
Bolboschoenus medianus	Endemic	New Zealand	Saltmarshes	V.J. Cook. (1947)
PHRYMACEAE				
Mimulus repens	Native	New Zealand, Australia		Auckland Regional Authority (1983)
RANUNCULACEAE				
Ranunculus acaulis	Native	New Zealand, Australia, Chile, Falkland Islands		Auckland Regional Authority (1983)
ZOSTERACEAE				
Zostera muelleri	Endemic	New Zealand		Chapman & Ronaldson (1958)
POACEAE				
Spartina alterniflora	Non-indigenous	Eastern U.S., Bay of Biscay, France, New Zealand	Tidal flats	Partridge, T.R. (1987)
Spartina townsendii	Non-indigenous	Britain, Belgian, Denmark, France, Ireland, New Zealand	Tidal saltmarsh	Partridge, T.R. (1987)
Species	Endemism	Distribution	Habitat	Source

Spartina townsendii var. anglica	Non-indigenous	U.K., Alaska, Sweden, Germany, Australia, New Zealand	Tidal saltmarsh	Halliday et al. (2006)
ACANTHACEAE				
Avicennia marina australasica	Native	New Zealand, S. Australia	Intertidal Mudflats	Hayward et al. (1997b)
Avicennia marina	Native	New Zealand, South Australia, Indian Ocean, Red Sea, South Africa, S. / SE Asia, S. Pacific Ocean, Japan, China, Spain	Intertidal Mudflats	Chapman & Ronaldson (1958)
JUNCACEAE				
Juncus maritimus	Native	NW European waters, New Zealand		Chapman & Ronaldson (1958)
AMARANTHACEAE				
Sarcocornia quinqueflora	Native	New Zealand, Australia		Hayward et al. (1997b)
Phylum PORIFERA				
Class CALCAREA				
CLATHRINIDAE				
Clathrina coriacea	Non-indigenous	N. Atlantic, Mediterranean, Japan, Indian Ocean, Arctic, Antarctic, European waters, New Zealand		Halliday et al. (2006)
LEUCASCIDAE				
Ascaltis poterium	Non-indigenous	Red Sea, Australia, New Zealand		Halliday et al. (2006)
Species	Endemism	Distribution	Habitat	Source
Class DEMOSPONGIAE				
CLIONAIDAE				

Cliona celata	Non-indigenous	Cosmopolitan		Halliday et al. (2006)
HALICHONDRIIDAE				
Halichondria (Halichondria) panicea	Non-indigenous	European waters, Brazil, New Zealand, Artic waters		Halliday et al. (2006)
Hymeniacidon perlevis	Non-indigenous	European waters, Brazil, New Zealand, Republic of the Congo		Halliday et al. (2006)
HALISARCIDAE				
Halisarca dujardinii	Non-indigenous	European waters, South Africa, Australia, New Zealand		Halliday et al. (2006)
TETHYIDAE				
Tethya aurantium	Non-indigenous	California, New Zealand, European waters		Halliday et al. (2006)
Tethya burtoni	Endemic	New Zealand		Shears, N. T. (2012)
CHALINIDAE				
Haliclona (Rhizoniera) rosea	Non-indigenous	European waters, Iceland, New Zealand		Halliday et al. (2006)
Haliclona parietalioides	Endemic	New Zealand		Inglis et al. (2006)
Haliclona venustina	Endemic	New Zealand		Inglis et al. (2006)
Haliclona maxima	Endemic	New Zealand		Inglis et al. (2006)
Haliclona stelliderma	Endemic	New Zealand		Inglis et al. (2006)
Haliclona n Sp. 1	Cryptogenic			Inglis et al. (2006)
Haliclona n Sp. 2	Cryptogenic			Inglis et al. (2006)
Haliclona n Sp. 3	Cryptogenic			Inglis et al. (2006)
Species	Endemism	Distribution	Habitat	Source
Haliclona n Sp. 4	Cryptogenic			Inglis et al. (2006)
Haliclona n Sp. 5	Cryptogenic			Inglis et al. (2006)

DYSIDEIDAE				
Euryspongia n. sp. 1	Cryptogenic			Inglis et al. (2006)
Euryspongia n. sp. 2	Cryptogenic			Inglis et al. (2006)
CALLYSPONGIIDAE				
Callyspongia ramosa	Cryptogenic	New Zealand, Australia		Inglis et al. (2006)
ESPERIOPSIDAE				
Esperiopsis Sp.	Cryptogenic			Inglis et al. (2006)
Class HOMOSCLEROMORPHA				
PLANKINIDAE				
Plakina monolopha	Non-indigenous	Mediterranean, Celtic Sea, New Zealand		Halliday et al. (2006)
Phylum CNIDARIA				
Class ANTHOZOA				
ACTINIIDAE				
Anthopleura aureoradiata	Native	New Zealand, SW Pacific		Grenfell & Hayward (1995)
DIADUMENIDAE				
Diadumene lineata	Non-indigenous	Cosmopolitan	Brackish tidal flats	Dromgoole & Foster (1983)
Species	Endemism	Distribution	Habitat	Source
CARYOPHYLLIIDAE				
Hoplangia durotrix	Non-indigenous	European waters, Mediterranean, SW Pacific		Halliday et al. (2006)

Tethocyathus cylindraceus	Non-indigenous	Gulf of Mexico, Bermuda, W. Atlantic, New Zealand		Halliday et al. (2006)
Class HYDROZOA				
CORYNIDAE				
Coryne pusilla	Non-indigenous	European waters, Mediterranean, Japan, New Zealand		Halliday et al. (2006)
TUBULARIIDAE				
Ectopleura crocea	Non-indigenous	NE U.S., European waters, Japan, NE Pacific, South Africa, New Zealand, Mediterranean, Gulf of Mexico, Canada, Australia, Alaska, Israel		Halliday et al. (2006)
EUDENDRIIDAE				
Eudendrium ritchiei	Non-indigenous	South Africa, New Zealand		Halliday et al. (2006)
PENNARIIDAE				
Pennaria disticha	Non-indigenous	Circum-global in warm water		Halliday et al. (2006)
BOUGAINVILLIIDAE				
Bougainvillia muscus	Non-indigenous	N. Atlantic, European waters, New Zealand, Gulf of Mexico		Halliday et al. (2006)
CAMPANULARIIDAE				
Clytia hemisphaerica	Non-indigenous	N. Atlantic, European waters, New Zealand, Gulf of Mexico, Mediterranean, Red Sea		Halliday et al. (2006)
Species	Endemism	Distribution	Habitat	Source

Obelia bidentata	Non-indigenous	N. Atlantic, European waters, New Zealand, Gulf of Mexico, Mediterranean, Red Sea, South America, South Carolina	Halliday et al. (2006)
Obelia longissima	Cryptogenic	New Zealand, Australia, European waters, Massachusetts	Inglis et al. (2006)
SERTULARIIDAE			
Amphisbetia fasciculata	Native	New Zealand, SW Pacific	Inglis et al. (2006)
SOLANDERIIDAE			
Solanderia ericopsis	Native	New Zealand, SW Pacific	Inglis et al. (2006)
Phylum ANNELIDA			
Class POLYCHAETA			
PARAONIDAE			
Aricidea sp.	Native		Cummings et al. (2002)
Levinsenia gracilis	Native	N. Atlantic, European waters, Greece, Caribbean, N. Canada, Cuba, Gulf of Mexico	Hewitt et al. (2006)
CAPITELLIDAE			
Heteromastus filiformis	Native	N. Pacific, California, Alaska, Caribbean, European waters, N. Atlantic, Mediterranean, Mozambique, New Zealand	Cummings et al. (2002)
Capitella Sp.	Native		Cummings et al. (2002)
SPIONIDAE			
Aonides oxycephala	Native	New Zealand, N. Atlantic, Japan, Mediterranean, Mozambique, Europe	Cummings et al. (2002)
Prionospio aucklandica	Endemic	New Zealand	Cummings et al. (2002)

Species	Endemism	Distribution	Habitat	Source
Scolecolepides sp.	Endemic	New Zealand		Cummings et al. (2002)
Scolelepis SP.	Endemic	New Zealand		Cummings et al. (2002)
Polydora cornuta	Non-indigenous	California, European waters, China Sea, Australia, New Zealand, Japan, Caribbean, Gulf of Mexico		Read & Gordon (1991)
Polydora haswelli	Non-indigenous	Australia, New Zealand		Halliday et al. (2006)
Pseudopolydora corniculata	Non-indigenous	Taiwan, New Zealand		Halliday et al. (2006)
COSSURIDAE				
Cossura consimilis	Endemic	New Zealand		Hewitt et al. (2006)
Boccardia Sp.	Unknown			Nicholls et al. (2002)
ORBINIIDAE				
Leodamas cirratus	Native	Chile, New Zealand		Cummings et al. (2002)
Scoloplos cylindrifer	Endemic	New Zealand		Cummings et al. (2002)
Phylo novazealandiae	Endemic	New Zealand		Cummings et al. (2002), Inglis et al. (2006)
Orbinia papillosa	Endemic	New Zealand		Cummings et al. (2002)
MALDANIDAE				
Macroclymenella stewartensis	Endemic	New Zealand		Cummings et al. (2002)
PECTINARIIDAE				
Lagis australis	Endemic	New Zealand		Cummings et al. (2002)
MAGELONIDAE				
Magelona dakini	Endemic	New Zealand		Cummings et al. (2002)

Species	Endemism	Distribution	Habitat	Source
NEPHTYIDAE				
Aglaophamus macroura	Endemic	New Zealand		Cummings et al. (2002)
Aglaophamus verrilli	Native	New Zealand, Venezuela, NW Atlantic, Gulf of Mexico, Caribbean		Inglis et al. (2006)
POLYNOIDAE				
Paralepidonotus ampulliferus	Native	New Zealand, Kenya, Mozambique, Red Sea		Read, G. B. (2006)
Harmothoe macrolepidota	Endemic	New Zealand		Inglis et al. (2006)
Lepidonotus polychromus	Endemic	New Zealand		Inglis et al. (2006)
SABELLIDAE				
Sabella spallanzanii	Non-indigenous	New Zealand, Australia, NW Atlantic, Mediterranean		Geoffrey et al. (2011)
Euchone Sp.	Native			Nicholls et al. (2002)
Megalomma suspiciens	Endemic	New Zealand		Inglis et al. (2006)
Pseudopotamilla alba	Endemic	New Zealand		Inglis et al. (2006)
Pseudopotamilla laciniosa	Endemic	New Zealand		Inglis et al. (2006)
SABELLARIIDAE				
Neosabellaria kaiparaensis	Endemic	New Zealand		Inglis et al. (2006)
Paraidanthyrsus quadricornis	Endemic	New Zealand		Inglis et al. (2006)
SERPULIDAE				
Spirobranchus cariniferus	Endemic	New Zealand		Dromgoole & Foster (1983)
Hydroides norvegica	Non-indigenous	New Zealand, European waters, Mediterranean		Dromgoole & Foster (1983)

Species	Endemism	Distribution	Habitat	Source
Hydroides elegans	Non-indigenous	European waters, Taiwan, Mediterranean, Japan, Gulf of Mexico, N. Atlantic, New Zealand, Argentina, South Africa, California		Halliday et al. (2006)
Ficopomatus enigmaticus	Non-indigenous	European waters, Hawaii, Mediterranean, Japan, Gulf of Mexico, N. Atlantic, New Zealand, Florida, Mexico, Caribbean		Read & Gordon (1991)
Hydroides ezoensis	Non-indigenous	Japan, European waters, Australia, New Zealand		Halliday et al. (2006)
Filograna implexa	native	New Zealand, Gulf of Mexico, European waters, Red Sea, Mozambique, Greece		Inglis et al. (2006)
CIRRATULIDAE				
Protocirrineris nuchalis	Endemic	New Zealand		Inglis et al. (2006)
Timarete anchylochaeta	Endemic	New Zealand		Inglis et al. (2006)
CHAETOPTERIDAE				
Chaetopterus Sp.	Non-indigenous			Halliday et al. (2006)
GLYCERIDAE				
Glycera lamelliformis	Endemic	New Zealand		Inglis et al. (2006)
LUMBRINERIDAE				
Lumbrineris sphaerocephala	Endemic	New Zealand		Inglis et al. (2006)
Ninoe leptognatha	Native	Gulf of Mexico, New Zealand		Sivaguru & Grace (2001)
GONIADIDAE				
Glycinde dorsalis	Endemic	New Zealand		Inglis et al. (2006)

Species	Endemism	Distribution	Habitat	Source
NEREIDIDAE				
Perinereis pseudocamiguina	Endemic	New Zealand		Inglis et al. (2006)
PHYLLODOCIDAE				
Eulalia microphylla	Endemic	New Zealand		Inglis et al. (2006)
SIGALIONIDAE				
Labiosthenolepis laevis	Endemic	New Zealand		Inglis et al. (2006)
SYLLIDAE				
Haplosyllis spongicola	native	New Zealand, South Africa, Red Sea, N. Atlantic, Mediterranean, Mexico, Gulf of Mexico, Caribbean, Cuba, Panama, Mozambique		Inglis et al. (2006)
Trypanosyllis zebra	native	New Zealand, South Africa, Red Sea, European waters, Mediterranean, Madagascar, Gulf of Mexico, Caribbean, Cuba		Inglis et al. (2006)
ACROCIRRIDAE				
Acrocirrus trisectus	Endemic	New Zealand		Inglis et al. (2006)
FLABELLIGERIDAE				
Flabelligera affinis	native	Mediterranean, European waters, N. Atlantic, N. Pacific, Canada		Inglis et al. (2006)
Daylithos parmatu	native	New Zealand, Madagascar, Colombia, Caribbean Sea, Philippines		Inglis et al. (2006)

Species	Endemism	Distribution	Habitat	Source
TEREBELLIDAE				
Pseudopista rostrata	Endemic	New Zealand		Inglis et al. (2006)
Streblosoma toddae	Endemic	New Zealand		Inglis et al. (2006)
Phylum MOLLUSCA				
Class GASTROPODA				
AMPHIBOLIDAE				
Amphibola crenata	Endemic	New Zealand	Intertidal, mudflats	Sivaguru & Grace (2001)
MURICIDAE				
Haustrum scobina	Endemic	New Zealand	Rocky Intertidal	Stewart et al. (1992)
Murexsul octogonus	Endemic	New Zealand		Powell, A. W. B. (1937)
Xymenella pusilla	Endemic	New Zealand		Hayward et al. (1997b)
Xymene plebeius	Endemic	New Zealand		Powell, A. W. B. (1937)
Dicathais orbita	Native	New Zealand, Australia, Lord Howe, Norfolk	Rocky Intertidal	
VOLUTIDAE				
Alcithoe arabica	Endemic	New Zealand		Hayward et al. (1997b)
CALYPTRAEIDAE				
Maoricrypta costata	Endemic	New Zealand		Hayward et al. (1997b)
Maoricrypta monoxyla	Endemic	New Zealand		Powell, A. W. B. (1937)
Sigapatella novaezelandiae	Endemic	New Zealand		Powell, A. W. B. (1937)
Sigapatella tenuis	Endemic	New Zealand		Powell, A. W. B. (1937)

Species	Endemism	Distribution	Habitat	Source
CALLIOSTOMATIDAE				
Calliostoma pellucidum	Endemic	New Zealand		Powell, A. W. B. (1937)
OLIVIDAE				
Amalda australis	Endemic	New Zealand		Powell, A. W. B. (1937)
Amalda mucronata	Endemic	New Zealand		Powell, A. W. B. (1937)
Amalda novaezelandiae	Endemic	New Zealand		Powell, A. W. B. (1937)
BUCCINIDAE				
Buccinulum linea	Endemic	New Zealand		Powell, A. W. B. (1937)
Cominella adspersa	Endemic	New Zealand		Powell, A. W. B. (1937)
Cominella quoyana	Endemic	New Zealand		Powell, A. W. B. (1937)
Penion sulcatus	Endemic	New Zealand		Powell, A. W. B. (1937)
Cominella glandiformis	Endemic	New Zealand		Sivaguru & Grace (2001)
Cominella maculosa	Endemic	New Zealand		Shears, N. T. (2012)
Cominella virgata	Endemic	New Zealand		Shears, N. T. (2012)
Buccinulum vittatum	Endemic	New Zealand		Shears, N. T. (2012)
EPITONIIDAE				
Epitonium scalare	Native	New Zealand, South Africa, Madagascar, Red Sea		Hayward et al. (1997b)
TURRITELLIDAE				
Maoricolpus roseus	Endemic	New Zealand, Non-indigenous in Australia		Powell, A. W. B. (1937)

Species	Endemism	Distribution	Habitat	Source
MANGELIIDAE				
Neoguraleus murdochi	Endemic	New Zealand		Powell, A. W. B. (1937)
Neoguraleus sinclairi	Endemic	New Zealand		Powell, A. W. B. (1937)
NATICIDAE				
Proxiuber australe	Endemic	New Zealand		Powell, A. W. B. (1937)
STRUTHIOLARIIDAE				
Pelicaria vermis	Endemic	New Zealand		Powell, A. W. B. (1937)
CAPULIDAE				
Trichosirius inornatus	Endemic	New Zealand		Powell, A. W. B. (1937)
TROCHIDAE				
Coelotrochus tiaratus	Endemic	New Zealand		Powell, A. W. B. (1937)
Diloma subrostratum	Endemic	New Zealand		Cummings et al. (2002)
Diloma aethiops	Endemic	New Zealand		Grange, K. R. (1976)
Diloma zelandicum	Endemic	New Zealand		Grange, K. R. (1976)
Coelotrochus viridis	Endemic	New Zealand		Shears, N. T. (2012)
Micrelenchus purpureus	Endemic	New Zealand		Shears, N. T. (2012)
Micrelenchus huttonii	Endemic	New Zealand		Sivaguru & Grace (2001)
LOTTIIDAE				
Notoacmea elongata	Endemic	New Zealand		Sivaguru & Grace (2001)
Notoacmea parviconoidea	Endemic	New Zealand		Suter, H. (1907)
Notoacmea daedala	Endemic	New Zealand		Suter, H. (1907)

Species	Endemism	Distribution	Habitat	Source
HAMINOEA				
Haminoea zelandiae	Endemic	New Zealand		Nicholls et al. (2002)
BATILLARIIDAE				
Zeacumantus lutulentus	Endemic	New Zealand		Cummings et al. (2002)
NASSARIIDAE				
Tritia burchardi	Endemic	New Zealand		Townsend et al. (2010)
TURBINIDAE				
Lunella smaragda	Endemic	New Zealand		Grange, K. R. (1976)
POLYCERIDAE				
Thecacera pennigera	Non-indigenous	New Zealand, Australia, European waters		Willan, R. C. (1976) Dromgoole & Foster (1983)
Polycera hedgpethi	Non-indigenous	New Zealand, Italy, Spain, Caribbean		Halliday et al. (2006)
NACELLIDAE				
Cellana radians	Endemic	New Zealand		Luckens, P. A. (1974)
GONIODORIDIDAE				
Okenia eolida	Non-indigenous	Cosmopolitan		Willan & Morton (1984) Hine, M (1995)
Okenia pellucida	Non-indigenous	Australia, New Zealand		Halliday et al. (2006)

Species	Endemism	Distribution	Habitat	Source
ELLOBIIDAE				
Microtralia Sp. (insularis)	Non-indigenous	Hong Kong, New Zealand		Halliday et al. (2006)
TERGIPEDIDAE				
Trinchesia alpha	Non-indigenous	Japan, New Zealand		Halliday et al. (2006)
Cuthona beta	Non-indigenous	Japan, New Zealand		Halliday et al. (2006)
EUBRANCHIDAE				
Eubranchus agrius	Non-indigenous	Chile, New Zealand		Halliday et al. (2006)
LITTORINIDAE				
Risellopsis varia	Endemic	New Zealand		Inglis et al. (2006)
ONCHIDELLA				
Onchidella nigricans	Endemic	New Zealand		Inglis et al. (2006)
FISSURELLIDAE				
Tugali suteri	Endemic	New Zealand		Inglis et al. (2006)
DENDRODORIDIDAE				
Dendrodoris citrina	Endemic	New Zealand		Shears, N. T. (2012)
Class POLYPLACOPHORA				
ACANTHOCHITONIDAE				
Acanthochitona mariae	Unknown			Powell, A. W. B. (1937)

Species	Endemism	Distribution	Habitat	Source
LEPTOCHITONIDAE				
Leptochiton inquinatus	Endemic	New Zealand		Powell, A. W. B. (1937)
ACANTHOCHITONIDAE				
Pseudotonicia cuneata	Endemic	New Zealand		Hayward et al. (1997b)
Acanthochitona zelandica	Endemic	New Zealand		Inglis et al. (2006)
Cryptoconchus porosus	Endemic	New Zealand		Inglis et al. (2006)
CHITONIDAE				
Rhyssoplax stangeri	Endemic	New Zealand		Hayward et al. (1997b)
Sypharochiton pelliserpentis	Endemic	New Zealand		Luckens, P. A. (1974)
Onithochiton neglectus	Endemic	New Zealand		Inglis et al. (2006)
Sypharochiton sinclairi	Endemic	New Zealand		Inglis et al. (2006)
Chiton glaucus	Native	New Zealand, Tasmania		Sivaguru & Grace (2001)
Class BIVALVIA				
ANOMIIDAE				
Anomia trigonopsis	Endemic	New Zealand		Powell, A. W. B. (1937)
PINNIDAE				
Atrina zelandica	Endemic	New Zealand		Page, R. D. M. (1983)
VENERIDAE				
Austrovenus stutchburyi	Endemic	New Zealand		Hayward et al. (1997b)
Bassina yatei	Endemic	New Zealand		Hayward et al. (1997b)
Dosina mactracea	Endemic	New Zealand		Powell, A. W. B. (1937)

Species	Endemism	Distribution	Habitat	Source
Dosinia greyi	Endemic	New Zealand		Powell, A. W. B. (1937)
Dosinia lambata	Endemic	New Zealand		Powell, A. W. B. (1937)
Dosinia maoriana	Endemic	New Zealand		Hayward et al. (1997b)
Dosinia subrosea	Endemic	New Zealand		Hayward et al. (1997b)
Venerupis largillierti	Endemic	New Zealand, Non-indigenous in Australia		Powell, A. W. B. (1937)
Tawera spissa	Endemic	New Zealand		Powell, A. W. B. (1937)
LASAEIDAE				
Borniola reniformis	Endemic	New Zealand		Powell, A. W. B. (1937)
Arthritica bifurca	Endemic	New Zealand		Cummings et al. (2002)
Lasaea hinemoa	Endemic	New Zealand		Inglis et al. (2006)
PECTINIDAE				
Talochlamys zelandiae	Endemic	New Zealand		Powell, A. W. B. (1937)
CORBULIDAE				
Corbula zelandica	Endemic	New Zealand		Powell, A. W. B. (1937)
MACTRIDAE				
Cyclomactra ovata	Endemic	New Zealand		Powell, A. W. B. (1937)
Zenatia acinaces	Endemic	New Zealand		Powell, A. W. B. (1937)
UNGULINIDAE				
Zemysia striatula	Endemic	New Zealand		Hayward et al. (1997b)
Zemysia zelandica	Endemic	New Zealand		Powell, A. W. B. (1937)

Species	Endemism	Distribution	Habitat	Source
PSAMMOBIIDAE				
Gari stangeri	Endemic	New Zealand		Hayward et al. (1997b)
Hiatula nitida	Endemic	New Zealand		Hayward et al. (1997b)
Hiatula siliquens	Endemic	New Zealand		Cummings et al. (2002)
HIATELLIDAE				
Hiatella arctica	Native	Arctic, Canada, Caribbean, N. Atlantic, European water, Mediterranean, Gulf of Mexico, British Isles		Hayward et al. (1997b)
SEMELIDAE				
Leptomya retiaria	Endemic	New Zealand		Powell, A. W. B. (1937)
Theora lubrica	Non-indigenous	Japan, tropical Pacific, Indonesia, Thailand, China, Philippines, Australia, Mediterranean, New Zealand	Soft benthic sediment	Dromgoole & Foster (1983)
LIMIDAE				
Limaria orientalis	Non-indigenous	Japan, New Zealand	Benthic shell gravel	Dromgoole & Foster (1983)
TELLINIDAE				
Macomona liliana	Endemic	New Zealand		Hayward et al. (1997b)
Bartschicoma edgari	Endemic	New Zealand		Hayward et al. (1997b)
MYTILIDAE				
Musculus impactus	Endemic	New Zealand		Hayward et al. (1997b)
Arcuatula senhousia	Non-indigenous	East Asia, Australia, European waters, Mediterranean, Kenya, New Zealand, India, California	Tidal flats	Willan, R. C. (1985)

Species	Endemism	Distribution	Habitat	Source
Perna canaliculus	Endemic	New Zealand		Powell, A. W. B. (1937)
Zelithophaga truncata	Endemic	New Zealand		Hayward et al. (1997b)
Modiolus areolatus	Endemic	New Zealand		Inglis et al. (2006)
Xenostrobus pulex	Native	New Zealand, Australia		Inglis et al. (2006)
MALLETIIDAE				
Neilo australis	Endemic	New Zealand		Powell, A. W. B. (1937)
NUCULIDAE				
Linucula hartvigiana	Endemic	New Zealand		Powell, A. W. B. (1937)
Nucula nitidula	Endemic	New Zealand		Powell, A. W. B. (1937)
MESODESMATIDAE				
Paphies australis	Endemic	New Zealand		Hayward et al. (1997b)
CARDITIDAE				
Pleuromeris zelandica	Endemic	New Zealand		Powell, A. W. B. (1937)
Purpurocardia purpurata	Endemic	New Zealand		Powell, A. W. B. (1937)
OSTREIDAE				
Ostrea chilensis	Native	New Zealand, U.K., France		Hayward et al. (1997b)
Crassostrea gigas	Non-indigenous	Cosmopolitan in temperate waters	Intertidal, hard substrate, tidal flats	Dinamani, P. (1971) Dromgoole & Foster (1983)
Saccostrea cucullata glomerata	Native			

Species	Endemism	Distribution	Habitat	Source
TEREDINIDAE				
Lyrodus medilobatus	Non-indigenous	Tropical Cosmopolitan		Halliday et al. (2006)
Lyrodus pedicellatus	Non-indigenous	Tropical to temperate seas, Cosmopolitan		Halliday et al. (2006)
ELLOBIIDAE				
Leuconopsis obsoleta	Endemic	New Zealand		Inglis et al. (2006)
Phylum ARTHROPODA				
Class PYCNOGONIDA				
AMMOTHEIDAE				
Achelia assimilis	Native	New Zealand, Australia, Campbell Plateau, South America, West Pacific		Inglis et al. (2006)
Class MALACOSTRACA				
PAGURIDAE				
Lophopagurus (Australeremus) Sp.	Unknown			Hayward et al. (1997b)
HYMENOSOMATIDAE				
Halicarcinus varius	Endemic	New Zealand		Hayward et al. (1997b)
Halicarcinus whitei	Endemic	New Zealand		Cummings et al. (2002)
Halicarcinus cookii	Endemic	New Zealand		Inglis et al. (2006)
Halimena aotearoa	Endemic	New Zealand		Inglis et al. (2006)
VARUNIDAE				
Austrohelice crassa	Endemic	New Zealand		Powell, A. W. B. (1937)
Hemigrapsus crenulatus	Native	New Zealand, French Polynesian		Cummings et al. (2002)

Species	Endemism	Distribution	Habitat	Source
MACROPHTHALMIDAE				
Hemiplax hirtipes	Endemic	New Zealand		Powell, A. W. B. (1937)
MAJIDAE				
Notomithrax minor	Endemic	New Zealand		Powell, A. W. B. (1937)
DIOGENIDAE				
Areopaguristes pilosus	Endemic	New Zealand		Hayward et al. (1997b)
PAGURIDAE				
Pagurus sp.	Unknown			Hayward et al. (1997b)
PORCELLANIDAE				
Petrolisthes novaezelandiae	Endemic	New Zealand		Powell, A. W. B. (1937)
Petrolisthes elongatus	Native	New Zealand, Australia		Inglis et al. (2006)
PILUMNIDAE				
Pilumnus novaezealandiae	Endemic	New Zealand		Powell, A. W. B. (1937)
Pilumnopeus serratifrons	Non-indigenous	South Australia		Halliday et al. (2006)
Pilumnus lumpinus	Endemic	New Zealand		Inglis et al. (2006)
PINNOTHERIDAE				
Nepinnotheres atrinicola	Endemic	New Zealand		Hayward et al. (1997b)
Nepinnotheres novaezelandiae	Endemic	New Zealand		Page, R. D. M. (1983)

Species	Endemism	Distribution	Habitat	Source
INACHOIDIDAE				
Pyromaia tuberculata	Non-indigenous	West coast N. America, New Zealand	Shallow subtidal soft sediment	Dromgoole & Foster (1983)
PORTUNIDAE				
Charybdis japonica	Non-indigenous	Japan, Korea, N. China, Malaysia		Smith et al. (2003)
PHOXOCEPHALIDAE				
Torridoharpinia hurleyi	Native	New Zealand, Southern Ocean		Cummings et al. (2002)
ALPHEOIDEA				
Alpheus richardsoni	Endemic	New Zealand		Cummings et al. (2002), Inglis et al. (2006)
DIASTYLIDAE				
Colurostylis lemurum	Endemic	New Zealand		Cummings et al. (2002)
POLYBIIDAE				
Ovalipes catharus	Endemic	New Zealand		Gust & Inglis (2004)
CHELURIDAE				
Chelura terebrans	Non-indigenous	Cosmopolitan		Chilton, C. (1914)
LIMNORIIDAE				
Limnoria lignorum	Non-indigenous	European waters, N. Pacific, Canada, Greece		Chilton, C. (1914)
Limnoria tripunctata	Non-indigenous	Cosmopolitan warm to tropical waters		Hayward, B. W. (1997)

Species	Endemism	Distribution	Habitat	Source
COROPHIIDAE				
Corophium sp. (contractum)	Unknown			Chilton, C. (1914)
Apocorophium acutum	Non-indigenous	Coasts of Europe		Halliday et al. (2006)
MELITIDAE				
Melita festiva	Endemic	New Zealand		Chilton, C. (1915)
SPHAEROMATIDAE				
Exospheroma Spp.	Unknown			Nicholls et al. (2002)
ISCHYROCERIDAE				
Ericthonius pugnax	Non-indigenous	SE Asia, Australia, Japan, Madagascar, New Zealand, Sri Lanka		Halliday et al. (2006)
LEUCOSIIDAE				
Merocryptus lambriformis	Non-indigenous	S. Australia, New Zealand		Halliday et al. (2006)
PLAGUSIIDAE				
Guinusia chabrus	Non-indigenous	Chile, Australia, New Zealand, South Africa		Halliday et al. (2006)
LILJEBORGIIDAE				
Liljeborgia akaroica	Endemic	New Zealand		Inglis et al. (2006)
LYSIANASSIDAE				
Parawaldeckia vesca	Native	New Zealand, Subantarctic		Inglis et al. (2006)
Acontiostoma Sp.	Cryptogenic	Unknown		Inglis et al. (2006)

Species	Endemism	Distribution	Habitat	Source
PODOCERUS				
Podocerus cristatus	Native	New Zealand, Subantarctic, Antarctic Ocean, Caribbean Sea, Mozambique, South Africa, Venezuela		Inglis et al. (2006)
CRANGONIDAE				
Pontophilus australis	Unknown			Inglis et al. (2006)
PALAEMONIDAE				
Periclimenes yaldwyni	Endemic	New Zealand		Inglis et al. (2006)
ARCTURIDAE				
Neastacilla tuberculata	Native	New Zealand, Temperate Australasia		Inglis et al. (2006)
CIROLANIDAE				
Cirolana kokoru	Native	New Zealand, Temperate Australasia		Inglis et al. (2006)
Cirolana quechso	Native	New Zealand, Temperate Australasia		Inglis et al. (2006)
Natatolana rossi	Native	New Zealand, Temperate Australasia		Inglis et al. (2006)
SPHAEROMATIDAE				
Pseudosphaeroma campbellense	Native	New Zealand, Temperate Australasia		Inglis et al. (2006)
Class OSTRACODA				
TRACHYLEBERIDIDAE				
Arculacythereis Sp.	Unknown			Morley & Hayward (2007)
Trachyleberis zeacristata	Endemic	New Zealand		Morley & Hayward (2007)
Actinocythereis thomsoni	Unknown			Morley & Hayward (2007)

Species	Endemism	Distribution	Habitat	Source
BYTHOCYTHERIDAE				
Baltraella Sp.	Unknown			Morley & Hayward (2007)
THAEROCYTHERIDAE				
Bradleya Sp.	Unknown			Morley & Hayward (2007)
Quadracythere mediaruga	Endemic	New Zealand		Morley & Hayward (2007)
Quadracythere radizea	Endemic	New Zealand		Morley & Hayward (2007)
LEPTOCYTHERIDAE				
Callistocythere dorsotuberculata	Endemic	New Zealand		Morley & Hayward (2007)
Callistocythere neoplana	Endemic	New Zealand		Morley & Hayward (2007)
Callistocythere obtusa	Endemic	New Zealand		Morley & Hayward (2007)
Leptocythere lacustris	Endemic	New Zealand		Morley & Hayward (2007)
Swansonella novaezealandica	Endemic	New Zealand		Morley & Hayward (2007)
PARADOXOSTOMATIDAE				
Paradoxostoma Sp.	Unknown			Morley & Hayward (2007)
HEMICYTHERIDAE				
Caudites scopulicolus	Native	New Zealand, Australia		Morley & Hayward (2007)
Hermanites briggsi	Endemic	New Zealand		Morley & Hayward (2007)
Hemingwayella pumilio	Unknown			Morley & Hayward (2007)
Procythereis (Serratocythere) lyttletonensis	Endemic	New Zealand		Morley & Hayward (2007)
Waiparacythereis joanae	Endemic	New Zealand		Morley & Hayward (2007)

Species	Endemism	Distribution	Habitat	Source
CYTHERELLIDAE				
Cytherella hemipuncta	Endemic	New Zealand		Morley & Hayward (2007)
Cytherelloidea willetti	Endemic	New Zealand		Morley & Hayward (2007)
CYTHERETTIDAE				
Cytheretta Sp.	Unknown			Morley & Hayward (2007)
CYTHERIDAIDAE				
Cytheridea aoteana	Endemic	New Zealand		Morley & Hayward (2007)
CYTHERURIDAE				
Cytheropteron latiscalpum	Endemic	New Zealand		Morley & Hayward (2007)
Oculocytheropteron dividentum	Endemic	New Zealand		Morley & Hayward (2007)
Hemicytherura fereplana	Endemic	New Zealand		Morley & Hayward (2007)
Hemicytherura pentagona	Endemic	New Zealand		Morley & Hayward (2007)
Microcytherura (Loxocythere) crassa	Endemic	New Zealand		Morley & Hayward (2007)
Microcytherura (Loxocythere) hornibrooki	Endemic	New Zealand		Morley & Hayward (2007)
Semicytherura sericava	Endemic	New Zealand		Morley & Hayward (2007)
Hemicytherura delicatula	Endemic	New Zealand		Morley & Hayward (2007)
EUCYTHERIDAE				
Eucythere (Rotundracythere) mytila	Endemic	New Zealand		Morley & Hayward (2007)
Eucythere (Rotundracythere) gravepuncta	Endemic	New Zealand		Morley & Hayward (2007)

Species	Endemism	Distribution	Habitat	Source
PECTOCYTHERIDAE				
Keijia demissa	Native	New Zealand, Gulf of Mexico		Morley & Hayward (2007)
Mckenziartia Sp.	Unknown			Morley & Hayward (2007)
Munseyella modesta	Endemic	New Zealand		Morley & Hayward (2007)
Munseyella brevis	Endemic	New Zealand		Morley & Hayward (2007)
Munseyella punctata	Endemic	New Zealand		Morley & Hayward (2007)
Parakeijia thomi	Endemic	New Zealand		Morley & Hayward (2007)
LOXOCONCHIDAE				
Loxoconcha punctata	Endemic	New Zealand		Morley & Hayward (2007)
MACROCYPRIDIDAE				
Macromckenziea porcelanica	Native	New Zealand, Australia		Morley & Hayward (2007)
PONTOCYPRIDIDAE				
Maddocksella tumefacta	Unknown			Morley & Hayward (2007)
CANDONIDAE				
Tasmanocypris Sp.	Unknown			Morley & Hayward (2007)
XESTOLEBERIDIDAE				
Xestoleberis Sp.	Unknown			Morley & Hayward (2007)
NEOCYTHERIDEIDIDAE				
Copytus novaezealandiae	Endemic	New Zealand		Morley & Hayward (2007)

Species	Endemism	Distribution	Habitat	Source
Subclass OSTRACODA INCERTAE SEDIS				
Swansonites tumida	Endemic	New Zealand		Morley & Hayward (2007)
Swansonites aequa	Endemic	New Zealand		Morley & Hayward (2007)
Infraclass Cirripedia				
AUSTROBALANIDAE				
Austrominius modestus	Native	Global		Chapman & Ronaldson (1958)
Amphibalanus variegatus	Non-indigenous	Indo-Malaysia, Australia, Belgium, France, North Sea, New Zealand		Dromgoole & Foster (1983)
Amphibalanus amphitrite	Non-indigenous	Cosmopolitan warm temperate & tropical seas	Intertidal hard substrate	Dromgoole & Foster (1983)
Balanus trigonus	Non-indigenous	Cosmopolitan warm temperate & tropical seas		Dromgoole & Foster (1983)
CHTHAMALIDAE				
Chaemosipho columna	Endemic	New Zealand		Inglis et al. (2006)
Subclass COPEPODA				
EUTERPINIDAE				
Euterpina acutifrons	Native	New Zealand, N. Atlantic, Turkey, Gulf of Mexico, Colombia		Moreira & Mcnamara (1984)
Phylum BRYOZOA				
Class GYMNOLAEMATA				
WATERSIPORIDAE				
Watersipora arcuata	Non-indigenous	California, Galapagos Islands, Australia, Ecuador, Mexico, N. Pacific.		Skerman, T. M. (1960)

Species	Endemism	Distribution	Habitat	Source
Watersipora subtorquata	Non-indigenous	Brazil, Bermuda, Cape Verde Islands, Bay of Biscay, Australia, France, Mexico, New Zealand, Hawaii, South Africa, U.K.		Gordon & Matawari (1992)
NOLELLIDAE				
Anguinella palmata	Non-indigenous	Southern European waters, Australia, Belgium, Mexico, Netherlands, New Zealand, North Sea, Panama		Morton & Miller (1968)
AEVERRILLIIDAE				
Aeverrillia armata	Non-indigenous	New Zealand, Gulf of Mexico, NE Pacific		Morton & Miller (1968)
VESICULARIIDAE				
Amathia distans	Non-indigenous	West Atlantic, Red Sea, Mediterranean, Australia, Japan, Hawaii, China, Yellow Sea, New Zealand		Gordon & Matawari (1992)
Amathia gracilis	Non-indigenous	NE America, California, Belgium, Netherlands, Canada, France, Gulf of Mexico		Gordon & Matawari (1992)
Amathia imbricata	Non-indigenous	Netherlands, North Sea, New Zealand, Ireland, France, Greece		Gordon & Matawari (1992)
Amathia verticillata	Non-indigenous	Mediterranean, Japan, Australia, Mexico, Gulf of Mexico, Portugal, South Korea, Hawaii		Gordon & Matawari (1992)
Amathia chimonidesi	Non-indigenous			Gordon & Spencer-Jones (2013)
BUGULIDAE				
Bugulina flabellata	Non-indigenous	European waters, Australia, New Zealand		Skerman, T. M (1959)
Bugula neritina	Non-indigenous	Cosmopolitan		Skerman, T. M (1959)
Bugulina stolonifera	Non-indigenous	European waters, Australia, New Zealand, China, Japan, India, Chile, Brazil, Argentina, Arabian Sea, Hawaii, Panama, California		Harger, J. R. E. (1964)

Species	Endemism	Distribution	Habitat	Source
BUSKIIDAE				
Buskia socialis	Non-indigenous	European waters, New Zealand		Gordon & Matawari (1992)
Buskia nitens	Non-indigenous	European waters, New Zealand		Gordon & Matawari (1992)
ELECTRIDAE				
Conopeum seurati	Non-indigenous	European waters, Mediterranean, South Africa, New Zealand, Japan		Gordon & Matawari (1992)
Arbopercula tenella	Non-indigenous	Atlantic coast of Florida, Puerto Rico, Brazil, New Zealand		Gordon & Matawari (1992)
CRYPTOSULIDAE				
Cryptosula pallasiana	Non-indigenous	Cosmopolitan		Gordon & Matawari (1992)
SCHIZOPORELLIDAE				
Schizoporella errata	Non-indigenous	New Zealand, Australia, Samoa, W. U.S.		Harger, J. R. E. (1964)
CANDIDAE				
Tricellaria occidentalis	Non-indigenous	New Zealand, Australia		Harger, J. R. E. (1964)
Tricellaria porteri	Non-indigenous	Australia, Japan, New Zealand		Halliday et al. (2006)
Caberea rostrata	Endemic	New Zealand		Harger, J. R. E. (1964)
Caberea zelandica	Endemic	New Zealand		Gordon, D. P. (1967)
LEPRALIELLIDAE				
Celleporaria SP.	Non-indigenous			Halliday et al. (2006)

Species	Endemism	Distribution	Habitat	Source
BEANIIDAE				
Beania n.sp.	Unknown			Inglis et al. (2006)
Beania plurispinosa	Native	New Zealand, S. Australia		Gordon & Matawari (1992)
CHAPERIIDAE				
Chaperiopsis cervicornis	Endemic	New Zealand		Gordon & Matawari (1992)
Chaperia granulosa	Endemic	New Zealand		Gordon & Matawari (1992)
HIPPOPORIDRIDAE				
Odontoporella n.sp	Unknown			
ALCYONIDIIDAE				
Alcyonidium sp.	Unknown			Gordon & Matawari (1992)
SCRUPARIIDAE				
Scruparia ambigua	Native	Cosmopolitan		Gordon & Matawari (1992)
BITECTIPORIDAE				
Schizosmittina cinctipora	Endemic	New Zealand		Gordon & Matawari (1992)
PHIDOLOPORIDAE				
Rhynchozoon larreyi	Native	New Zealand, Sri Lanka, Indonesia, Turkey		Gordon & Matawari (1992)

Species	Endemism	Distribution	Habitat	Source
Phylum BRACHIOPODA				
Class RHYNCHONELLATA				
TEREBRATELLIDAE				
Calloria inconspicua	Endemic	New Zealand		Powell, A. W. B. (1937)
Phylum ECHINODERMATA				
Class ECHINOIDEA				
LOVENIIDAE				
Echinocardium cordatum	Native	Temperate Cosmopolitan		Powell, A. W. B. (1937)
CLYPEASTERIDAE				
Fellaster zelandiae	Endemic	New Zealand		Powell, A. W. B. (1937)
ECHINOMETRIDAE				
Evechinus chloroticus	Endemic	New Zealand		Shears, N. T. (2012)
Class ASTEROIDEA				
STICHASTERIDAE				
Allostichaster polyplax	Endemic	New Zealand		Hayward et al. (1997b)
ASTERIIDAE				
Coscinasterias calamaria	Native	New Zealand, Angola, Mascarene Basin, South Africa, Madagascar		Powell, A. W. B. (1937)
ASTERINIDAE				
Patiriella regularis	Endemic	New Zealand, Non-indigenous in Australia		Hayward et al. (1997b)

Species	Endemism	Distribution	Habitat	Source
Stegnaster inflatus	Endemic	New Zealand		Hayward et al. (1997b)
Class OPHIUROIDEA				
AMPHIURIDAE				
Ophiocentrus pilosa	Unknown			Powell, A. W. B. (1937)
Amphipholis squamata	Native	Cosmopolitan		Hayward et al. (1997b)
Amphiura (Amphiura) rosea	Endemic	New Zealand		Powell, A. W. B. (1937)
OPHIACTIDAE				
Ophiactis resiliens	Endemic	New Zealand		Hayward et al. (1997b)
Ophionephthys Perplexa	Unknown			Hayward et al. (1997b)
Phylum HEMICHORDATA				
Class ENTEROPNEUSTA				
PTYCHODERIDAE				
Balanoglossus australiensis	Native	New Zealand, SW Pacific		Morton, J. E. (1950)
Phylum CHORDATA				
Class ASCIDIACEA				
STYELIDAE				
Styela clava	Non-indigenous	East Asia, European waters, N. Atlantic, Australia, California		Hayward & Morley (2009)
Asterocarpa humilis	Non-indigenous	U.K., Australia, Sub-Antarctica, Chile		Halliday et al. (2006)
Botrylloides magnicoecum	Non-indigenous	French Guiana, South Africa, New Zealand		Halliday et al. (2006)
Botryllus schlosseri	Non-indigenous	N. Atlantic, N. Pacific, Gulf of Mexico, Australia, Canada, New Zealand, European waters, South Africa, China, California, Florida		Halliday et al. (2006)

Species	Endemism	Distribution	Habitat	Source
Styela plicata	Non-indigenous	Cosmopolitan		Halliday et al. (2006)
Botrylloides leachii	Non-indigenous	European waters, Mediterranean, Australia, New Zealand		Halliday et al. (2006)
Polyzoa opuntia	Native	New Zealand, Southern Ocean		Inglis et al. (2006)
Asterocarpa coerulea	Endemic	New Zealand		Inglis et al. (2006)
Cnemidocarpa bicornuta	Endemic	New Zealand		Inglis et al. (2006)
Cnemidocarpa nisiotis	Endemic	New Zealand		Inglis et al. (2006)
Cnemidocarpa Sp.	Non-indigenous			Inglis et al. (2006)
Polycarpa pegasis	Endemic	New Zealand		Inglis et al. (2006)
DIDEMNIDAE				
Didemnum candidum	Non-indigenous	Cosmopolitan		Halliday et al. (2006)
Diplosoma listerianum	Non-indigenous	Cosmopolitan		Halliday et al. (2006)
Lissoclinum notti	Native	New Zealand, SW Pacific		Inglis et al. (2006)
CIONIDAE				
Ciona intestinalis	Non-indigenous	European waters, N. Atlantic, Brazil, Argentina, Australia, New Zealand, East Asia, Chile, Hawaii, South Africa	Intertidal hard substrate	Dromgoole & Foster (1983)
CORELLIDAE				
Corella eumyota	Non-indigenous	Australia, South Africa, Southern Ocean, U.K., France, New Zealand		Halliday et al. (2006)
POLYCITORIDAE				
Cystodytes dellechiajei	Non-indigenous	Warm water Cosmopolitan		Halliday et al. (2006)
Aplidium phortax	Non-indigenous	NE Australia, Solomon Islands, New Zealand		Halliday et al. (2006)

Species	Endemism	Distribution	Habitat	Source
POLYCLINIDAE				
Aplidium adamsi	Endemic	New Zealand		Inglis et al. (2006)
MOLGULIDAE				
Molgula enodis	Native	New Zealand, Southern Ocean		Inglis et al. (2006)
Molgula mortenseni	Native	New Zealand, SW Pacific		Inglis et al. (2006)
PYURIDAE				
Microcosmus australis	Native	New Zealand, SW Pacific		Inglis et al. (2006)
Pyura cancellata	Endemic	New Zealand		Inglis et al. (2006)
Pyura picta	Endemic	New Zealand		Inglis et al. (2006)
Pyura rugata	Endemic	New Zealand		Inglis et al. (2006)
Pyura subuculata	Endemic	New Zealand		Inglis et al. (2006)
Microcosmus squamiger	Cryptogenic	Mediterranean, India, New Zealand, N. Atlantic, N. Pacific, South Africa, California		Inglis et al. (2006)
Class ACTINOPTERI				
GOBIIDAE				
Acentrogobius pflaumii	Non-indigenous	Japan, Korea, Taiwan, Philippines, Australia, New Zealand	Demersal	Francis et al. (2003)
Arenigobius bifrenatus	Non-indigenous	S. Australia, New Zealand	Demersal	Wilis et al. (1999)
Favonigobius lentiginosus	Native	New Zealand, Australia	Demersal	Francis et al. (2003)
Favonigobius exquisitus	Native	New Zealand, SE Australia	Demersal	Francis et al. (2003)
BLENNIIDAE				
Omobranchus anolius	Non-indigenous	S. Australia, New Zealand	Demersal	Halliday et al. (2006)

Species	Endemism	Distribution	Habitat	Source
ANGUILLIDAE				
Anguilla australis australis	Native	New Zealand, SE Australia, South Pacific Islands	Amphidromous, Pelagic	Francis et al. (2005)
CONGRIDAE				
Conger wilsoni	Native	New Zealand, S. Australia	Demersal	Inglis et al. (2006)
MUGILIDAE				
Aldrichetta forsteri	Native	New Zealand, S. Australia	Pelagic	Hurst et al. (2000)
Mugil cephalus	Native	Costal cosmopolitan	Pelagic	Morrison & Lowe (2012)
CARANGIDAE				
Decapterus koheru	Endemic	New Zealand	Pelagic	Inglis et al. (2006)
Trachurus novaezelandiae	Native	New Zealand, Australia	Pelagic	Inglis et al. (2006)
Seriola lalandi	Native	Southern Hemisphere waters	Pelagic	Larcombe, M.F. (1973)
Pseudocaranx dentex	Native	Tropical and warm temperate areas of the Atlantic, Mediterranean, Indian, and Pacific Oceans.	Pelagic	Larcombe, M.F. (1973)
LABRIDAE				
Notolabrus celidotus	Endemic	New Zealand	Pelagic	Inglis et al. (2006)
SPARIDAE				
Chrysophrys auratus	Native	Australasia, East Asia	Pelagic	Larcombe, M.F. (1973)
TRIPTERYGIIDAE				
Forsterygion lapillum	Endemic	New Zealand	Demersal	Inglis et al. (2006)

Species	Endemism	Distribution	Habitat	Source
Forsterygion capito	Endemic	New Zealand	Demersal	Morrison & Lowe (2012)
Grahamina nigripenne	Endemic	New Zealand	Demersal	Morrison & Lowe (2012)
Forsterygion malcolmi	Endemic	New Zealand	Demersal	Morrison & Lowe (2012)
GALAXIIDAE				
Galaxias maculatus	Native	Patagonia, Argentina, Falkland Islands, Pacific Islands, New Caledonia, Australia, New Zealand	Amphidromous Pelagic	McDowall, R.M. (1990)
ELEOTRIDAE				
Gobiomorphus gobioides	Endemic	New Zealand	Demersal	McDowall, R.M. (1990)
ARRIPIDAE				
Arripis trutta	Native	New Zealand, SE Australia	Pelagic	Larcombe, M.F. (1973)
ENGRAULIDAE				
Engraulis australis	Native	New Zealand, SE Australia	Pelagic	Morrison & Lowe (2012)
RETROPINNIDAE				
Retropinna retropinna	Endemic	New Zealand	Pelagic	Morrison & Lowe (2012)
PLEURONECTIDAE				
Rhombosolea leporina	Endemic	New Zealand	Demersal	Hurst et al. (2000)
Rhombosolea plebeia	Endemic	New Zealand	Demersal	Hurst et al. (2000)
Peltorhamphus latus	Native	New Zealand, Norfolk Islands	Demersal	Morrison & Lowe (2012)

Species	Endemism	Distribution	Habitat	Source
KYPHOSIDAE				
Girella tricuspidata	Native	New Zealand, SE Australia	Pelagic	Morrison & Lowe (2012)
CLUPEIDAE				
Sprattus spp.	Unknown		Pelagic	Morrison & Lowe (2012)
Class ELASMOBRANCHII				
TRIAKIDAE				
Mustelus lenticulatus	Endemic	New Zealand	Demersal	Hurst et al. (2005)
CARCHARHINIDAE				
Carcharhinus brachyurus	Native	Coastal temperate waters	Pelagic	Larcombe, M.F. (1973)
DASYATIDAE				
Dasyatis brevicaudata	Native	New Zealand, South Australia, South Africa	Demersal	Larcombe, M.F. (1973)
MYLIOBATIDAE				
Myliobatis tenuicaudatus	Native	New Zealand, Australia	Demersal	Larcombe, M.F. (1973)
Class AVES				
CHARADRIIDAE				
Anarhynchus frontalis	Endemic	New Zealand		Auckland Regional Authority (1983)
Charadrius obscurus aquilonius	Endemic	New Zealand		Auckland Regional Authority (1983)
Charadrius bicinctus	Endemic	New Zealand		Dowding & Moore (2006)

Species	Endemism	Distribution	Habitat	Source
SCOLOPACIDAE				
Calidris canutus	Native	W. Africa, Alaska, Greenland, NW European waters, Russia, South America, Brazil, Australasia, India, East Asia		Auckland Regional Authority (1983)
Limosa lapponica baueri	Native	East Asia, Alaska, Australasia, European waters, Africa		Auckland Regional Authority (1983)
THRESKIORNITHIDAE				
Platalea regia	Native	Australia, New Zealand, Papa New Guinea, Indonesia, Solomon Islands.	Tidal flats, wetlands, shallows	Auckland Regional Authority (1983)
HAEMATOPODIDAE				
Haematopus finschi	Endemic	New Zealand	Tidal flats, sand dunes	Dowding & Moore (2006)
Haematopus unicolor	Endemic	New Zealand	Tidal flats, sand dunes	Dowding & Moore (2006)
RECURVIROSTRIDAE				
Himantopus himantopus leucocephalus	Native	Australasia, Japan, SE Asia	Tidal flats, wetlands	Dowding & Moore (2006)
STERNIDAE				
Sterna striata	Native	New Zealand, Southern Australia		Dowding & Moore (2006)
Hydroprogne caspia	Native	N. America, Baltic Sea, Black Sea, Asia, Africa, New Zealand, Australia, Caribbean Basin		Dowding & Moore (2006)
PHALACROCORACIDAE				
Phalacrocorax carbo	Native	Australasia, European waters, South Africa, Faroe Islands, Iceland, Green land, NE America		Dowding & Moore (2006)
Phalacrocorax sulcirostris	Native	Australia, Northern New Zealand		Dowding & Moore (2006)

Species	Endemism	Distribution	Habitat	Source
Phalacrocorax varius	Native	Australasia		Dowding & Moore (2006)
RALLIDAE				
Porzana tabuensis	Native	Australasia, Philippines, South Pacific Islands, Micronesia		Auckland Regional Authority (1983)
Gallirallus philippensis	Native	Australasia, Philippines, Subantartic islands, South Pacific Islands		Auckland Regional Authority (1983)
LOCUSTELLIDAE				
Megalurus punctatus	Endemic	New Zealand	Wetlands	Auckland Regional Authority (1983)
Class MAMMALIA				
DELPHINIDAE				
Tursiops truncatus	Native	Cosmopolitan within tropical / temperate waters		Constantine et al. (2015)
Orcinus orca	Native	Cosmopolitan		Constantine et al. (2015)