Rahui And Marine Construction:

Potential For Enhancement of Taonga species

A thesis

Submitted in partial fulfilment

of the requirements for the degree of

Master of Science

Major in Biological Science

By

Danielle Fox

THE UNIVERSITY OF
WAIKATO
Te Whare Wananga o Waikato

Hamilton, New Zealand.

February 2010

Abstract

The aims of my study were to investigate whether marine reserves enhance intertidal species used by Māori in a traditional or contemporary sense, and whether artificial structures in the intertidal region (such as wharf and bridge pilings) provide suitable habitats for traditionally harvested species. Further, I investigated whether non-indigenous species were found in these habitats, which may affect traditionally used species.

The abundance of ataata (cat's eyes; *Turbo smaragdus*) and kina (sea urchin; *Evechinus chloroticus*) were quantified in three marine protected areas and nearby unprotected reference beaches. Results from Mann-Whitney *U* comparisons suggested that kina were significantly enhanced inside the two marine reserves with total protection. Kina are released from harvest pressure inside these reserves and possibly from the effects of predation and competition. In one reserve ataata abundance was significantly lower and no difference was found for ataata in the second reserve. This species is not as heavily harvested as kina and may even be negatively affected by trampling inside reserves. Both kina and ataata showed no response to a partially protected marine park. Abundance of both non-indigenous species was not sufficient enough to be sampled and statistically quantified at the marine reserve sites. Biotic resistance may restrict the proliferation of non-indigenous species in rocky intertidal marine reserves.

Intertidal fauna were scraped from concrete and wooden structures and were compared with fauna inhabiting nearby natural rocky reefs. Multi-dimensional scaling and ANOSIM was used to explore trends in the community composition of different study sites, and to illustrate which habitat types are more associated with traditionally used and non-indigenous species. The main results conclude that diverse communities are associated with natural habitats whereas constructed structures have limited fauna. Of the traditionally used species, the native rock oyster (*Saccostrea glomerata*) was found in relatively equal frequencies on artificial habitats and rocky reefs, and the non-indigenous oyster (*Crassostrea gigas*) was predominantly recorded on artificial structures and rare on natural substrates. The construction of artificial structures around New Zealand's coastline may assist in the spread of non-indigenous species, to the detriment of species traditionally used by Māori

Acknowledgements

First and most importantly my gratitude is to my Supervisor Dr. Ian Duggan (Pukenga of Biological Sciences), who has supported me throughout my studies and encouraged me to research coastal biology, which is an important area for Māori people. He has also encouraged and financially supported me to share my research with other indigenous people around the world, by sending me to Canada to attend and present my research at the 12th International Congress of Ethnobiology.

Thanks to Te Tipu Pūtaiao for funding this research. Their contribution made it possible for me to sample a variety of areas in the North Island.

My gratitude is also extended to Wiki Papa (Māori science support) who has also supported my studies and offered me a place where Māori can come together in a family-like environment. She has also played a major role in assisting my trip to Canada and so I'm thankful to her for that.

My biggest thanks go to my awesome research crew because there is no way I could have completed this without all of you.

To my partner, James Blackwell thanks for coming on every trip, counting all our kina and having to learn more than you would want to about rocky shores. Thanks to my sister Court for counting barnacles and oysters with me in the rain. I am also thankful to my mates Chrystal Wipaki, James Watson and Christina Couling for waking up way too early to drive hours so I can collect my samples. Thanks to Helen Blackwell for counting kina in Gisborne reserve. Thanks to my other sister Jessica for coming to Mahia to collect barnacles and thanks to my dad for building me a quadrat. This thesis is dedicated to my family, to my parents Michael and Louisa, my sisters, my brother Maia and my partner James. Everything is always for you.

Table of Contents

Chapter One: Introduction

Table of Contents

Table of Contents

List of Figures

List of Tables

INTRODUCTION

The indigenous Māori people of New Zealand have important connections with the land and ocean where each tribe assists in the management of their tribal area. Guardianship of resources is an important role that involves families and tribes commitment to maintain and manage an areas local resource. Tikanga is an intricate framework based on protocol, ritual and rules that controls any fishing or harvesting within a tribe (Booth *et al.* 2003). Harvesting of seafood (or kaimoana) is an important part of the culture where various techniques and locations of acquiring seafood is passed down through generations. One of the main sources of food came from gathering seafood during low tide in the sand or on rocky shores, and these food types are still important and consumed by Māori people today (Gibson, 2005). Māori used intertidal regions extensively for food in pre-European times. The introduction of fishing methods used after European settlement focused more on demersal subtidal species, because early European settlers considered intertidal species unsuitable for consumption (Horn *et al*, 1999). Most of the intertidal fishing methods have been lost and this is reiterated by the fact that Māori now only have names for 9% of intertidal fish compared to 63% of fish species that inhabit pelagic regions (Horn *et al.* 1999).

Protection of kaimoana from exploitation

The introductions of marine reserves, parks and rahui have provided a means of protecting native species that are preferentially harvested. Marine reserves were firstly established in New Zealand in 1975 at Goat Island near Leigh to help replenish New Zealand’s marine life, and since then a further 28 reserves have been established throughout the country. At present marine reserves account for only 5% of New Zealand’s coast and much of this percentage is situated around the Kermadec and Auckland offshore islands (Enderby & Enderby, 2006). Marine parks have also been created around New Zealand but have different regulations administered by the Ministry of Fisheries. Marine parks differ from marine reserves in that reserves offer a greater level of protection. Two marine parks are established in New Zealand and these include Tawharanui and Mimiwhangata Parks. The Mimiwhangata Marine Park has a complete ban on commercial fishing but does allow recreational fishing as well as harvesting of specific shellfish (DOC, 2007). Marine reserves in New Zealand have been shown to enhance the abundance of fish and crayfish species, which were traditionally used by Māori people (Gibson, 2005). However, research to date shows a clear bias toward the effects of these marine reserves on commercial or recreational species and pelagic dwelling species. This bias excludes the majority of traditional food sources that reside in the intertidal regions. A number of animals and macro-algae (seaweed) living on the rocky shore were traditionally, and are contemporarily, used as food, for fishing and other uses. For example, green-lipped mussels (kuku; *Perna canaliculus*) are important as it is a food source, but the shell was also used for cutting hair and fibres for flax weaving and for making traditional cloaks (korowai). These are now a contemporary resource sold in supermarkets in NZ and exported widely both as frozen meat and live. Paua species (abalone, hiwahiwa; *Haliotis spp.*) and Cat’s eye (Ataata; *Turbo smaragdus*) are used for food but their shell is used as decorations in jewellery. Kina (sea urchin; *Evechinus chloroticus*), while usually regarded as a pest in most countries due to their feeding preferences causing barrens, are commonly considered a delicacy in

New Zealand, and are a favourite food among Māori people. Bullkelp (rimurapa; *Durvillea antarctica*) and red seaweeds (parengo; *Porphyra spp.*) are also important food and preservative sources. These species may either be susceptible to the increased number of predators they will now be subjected to at high tide inside reserves, or alternatively may be able to compete and survive well in these reserves. The effects of marine reserves on these traditionally important species has not been investigated and they are not likely to become the main focus of any private or public sector research agencies, unless they have particular commercial or recreational value.

Complex procedures presided over the relationship between fishermen and the sea, because the ocean was more than just a source of food but it held important spiritual significance to Māori traditions (Sandrey, 1986). Māori have developed similar conservation techniques for protection of species to marine reserves, such as the practise of Rahui in order to conserve or enhance depleted resources. The primary difference is that a Rahui has a temporary ban, whereas the resources within a marine reserve remain permanently retired. Rahui were generally administered at certain times of the year, particularly for certain kaimoana that had prominent seasonal cycles (Sandrey, 1986). Several other fishery management tools are available for contemporary Māori to manage areas important for fisheries. Taiapure are recognised as traditionally important areas where Māori have input on management regimes, but where commercial operations can still take place, while maitaitai reserves are also areas of traditional importance, but where tribes manage and control the harvest of seafood for non-commercial purposes (Williams, 2006). The traditional practise of Rahui may today be sought by tribal people and applied within taiapure or maitaitai reserves, giving this practise a place in modern law. For example, important occasions such as birthdays, weddings and funerals are always celebrated with seafood in Māori culture and so permits need to be gained from elders (Kaitiaki) and the Ministry of Fisheries who look after the areas fishing grounds. The South Island tribe, Ngai Tahu, administered a rahui in the 1980s, and Pukerua Bay near Wellington and Karekare near Auckland are also under long-term Rahui. In both taiapure and maitaitai reserves, these regulations need approval from the Ministry of Fisheries, and apply to both Māori and non-Māori.

Prior to European settlement, each Māori tribe governed over their area and marked boundaries with traditional poles to signify territories for recognition to neighbouring tribes. So in a way Māori people have always employed a sense of conservation in that each guardian or chief was responsible for their own areas sustainability of food resources (Gibson, 2005). The practise of Rahui was undertaken long before fishing regulations were officially implemented, but the research to date lacks current scientific evidence relating to the traditional (and contemporary) practise of Rahui on intertidal species. Because the resources within reserves remain unavailable they can act as a reference for testing the biological effects for rahui sites, as we have numerous reserves in New Zealand with known ages since their establishment. Information about traditionally important species in reserves will provide evidence for the practice of rahui and give an indication of the appropriate duration rahui should be used.

Marine enhancement

Various methods have been attempted to help sustain and boost local fisheries, such as importing species from other countries and using various tools to enhance depleted populations (Booth & Cox 2003). Artificial habitats are also designed to enhance aquaculture, restoration, commercial and recreational fishing (Booth & Cox 2003). Artificial reefs have been shown to enhance subtidal populations of paua in Japan and it has been suggested that there is potential for reefs to enhance paua populations in New Zealand (Booth & Cox, 2003). However, it is not known if constructed structures enhance populations of traditionally used species in New Zealand.

Spread of non-indigenous species

Species dispersal is a natural biological mechanism required for various reasons such as migration pathways for food and mates or to simply find new homes. So why should we care about non-indigenous species that invade our native habitats? It is well recognised that the accidental or intentional introduction of non-indigenous species by human activities such as shipping, is one of the major causes for the decline in biodiversity globally (Stachowicz *et al.,* 1999). Species dispersal is usually restricted to specific temporal, spatial, and physical barriers depending on the species. However, shipping has provided the transport mechanism for which different organisms may overcome natural barriers and spread around the world. Up to tens of thousands of species of bacteria, plants and animals are being transported between oceans annually (Carlton, 1996). Organisms typically attach themselves to the ships hull or remain in the ballast. Regulations have since been put in place to try overcome the problem of organisms being discharged from ballast water. Due to the consequences concerned with invasive species in the North American Great Lakes, a ballast water regulation has been implemented in the USA. Encrusting organisms such as bryozoans and molluscs attach to a ships hull and pose a threat if they spawn upon arrival into foreign habitats. Previously anti-fouling paints were applied to boats to reduce the impact of hull fouling species. However, due to the poisonous chemical Tributyl-tin (TBT) in the hull-fouling paints, the International Maritime Organisation has developed a convention for the International Convention on the Control of Harmful Anti-fouling Systems on Ships (Gollasch, 2006). Cranfield *et al.* (2006) have compiled an extensive list of non-indigenous species in New Zealand and they have reported that the majority (~119 species) have arrived via hull fouling.

Features that determine an ecosystems invasibility

So what determines whether or not an ecosystem is likely to be invaded by non-indigenous species? And are there deterministic features of that ecosystem which causes them to be more susceptible to becoming invaded? In invasion ecology there are conflicting views concerning features that make an environment becoming invaded. Different features of an ecosystem (for example biotic and abiotic factors) will determine whether or not a non-indigenous species will spread and become established within the ecosystem. The effects that diversity has on resource use are considered to be the primary influence on a community's susceptibility, because there is a more complete use of resources and therefore less available for invasive species. This concept has been defined as the "diversity (or biotic) resistance hypothesis" by Levine (2000). This hypothesis has caused much debate amoung scientists as they have observed an invasion of non-indigenous species in habitats with both high and low biodiversity. These conflicting results are now thought to be due to differences in spatial scale. Biotic interactions may work on small spatial scales such as a diverse community resisting an invasion, but processes such as propagule supply, which may work on larger spatial scales, may overwhelm the defenses of a diverse community (Levine, 2000). Among different habitats there will be slightly different factors that may favour the susceptibility of an ecosystem. Many observational studies have reported non-indigenous species after they are already established, but manipulative experiments are required to understand the underlying processes and mechanisms which determine a communities resistance to invasion (Stachowicz *et al.* 1999). Diversity, resources and available space are the interacting factors thought to primarily affect invasion success in rocky shore areas. Romanuk and Kolasa (2005) tested the diversity theory through microcosm experiments that represented rock pool community assemblages. By controlling diversity and nutrient resources they were able to conclude that resource limitation was the controlling factor for the invasibility of rock pool communities. Through their experiments they were able to determine that in high nutrient microcosms, invasion was successful when there was low biodiversity, but unsuccessful when biodiversity was high. Through experimental manipulations Stachowicz *et al.* (1999) was able to show that as available space becomes occupied with native species, the number of surviving invasive species will simultaneously decrease.

Non-indigenous species and artificial structures

The introduction of non-native species into New Zealand's marine environment is a contemporary problem that may affect traditional resources in reserves and rahui sites. However, the interactions between culturally important species and non-indigenous species are unknown. Artificial structures in marine environments are necessary to provide ports for shipping activities and also for recreational purposes. This is a potential contemporary problem that may aid in the establishment of non-indigenous species (NIS) to the detriment of culturally important species. Studies have shown that invasive species tend to be more frequently associated with artificial structures in marine environments. Glasby *et al* (2006) investigated the small-scale spread of non-indigenous species after the arrival and introduction phase. Their work focused on habitat use and in particular, native species versus non-indigenous species with artificial and natural habitats. Their results revealed that non-indigenous species were more prolific on artificial habitats than were native species, and native species were more abundant on reefs and seawalls compared to non-indigenous species. This association with artificial structures may also occur as they represent new habitats that are not rapidly colonised. Native species have evolved to inhabit natural substrates such as rocky reefs, and therefore may not colonise newly constructed habitats.

Competition between indigenous and non-indigenous species

Space is a limiting resource in marine environments and so the creation of newly formed environments provides space for the recruitment of non-indigenous species (Glasby *et al.* 2007). Shinen *et al*. (2009) investigated the invasion resistance on rocky shores and the ability of native predators to restrict the invasion front of the exotic mussel *Mytilus galloprovincialis*. They studied the direct effects that predators had on two native mussels and the invasive *Mytilus galloprovincialis,* and found that predator pressure was the major influence restricting *Mytilus galloprovincialis* establishment and spread along the Californian open coastline, and furthermore the only place where the invasive mussel was dominant was in protected bays and harbours where predator access was restricted. Bays and harbours can therefore aid in the shelter and establishment of this invasive mussel and act as larval sources for open coast communities (Shinen *et al*. 2009). Artificial habitats are mainly constructed in harbours and bays where boats can be anchored. There is a potential for non-indigenous species to disperse along a coastline if subsequent artificial structures are created within these embayments.

Potential threats to New Zealand protected ecosystems

If non-indigenous species can spread to new habitats following initial establishment in a constructed habitat, this may have implications for marine reserves. Marine reserves are used as a management tool for preserving or protecting native species from harvest, thus allowing populations of species to replenish and ideally help restock depleted populations outside reserve areas. Within a marine reserve all species benefit from the prevention of harvesting and as an advantage, populations markedly increase in abundances and in some cases sizes. However, some species may not benefit from reserve status because marine reserves indirectly boost predator numbers, which in turn forces prey species to decrease in abundance. Menge (2000) gives a detailed explanation of ‘top-down’ control within intertidal communities, where different processes regulate communities and these then affect trophic levels further down the food chain. It would be expected that because marine reserves are so heavily populated with co-existing species that the resources should be partitioned and utilised efficiently. The ‘diversity resistance hypothesis’ (Levine, 2000) would therefore seem applicable in these ecosystems, and it should be difficult for non-indigenous species to invade. However, recent studies have revealed contrary results that indicate marine reserves are definitely not resistant to invasions. A non-native intertidal seaweed (*Sargassum muticum*) and oyster species (*Crassostrea gigas*) were found by Klinger *et al*. (2006) to be highly abundant within reserves compared to their abundance outside reserve areas in the San Juan Archipelago (USA). Klinger *et al.* (2006) have shown that populations of both species are viable and that multiple recruitment events have occurred since invasion. This represents a huge problem for managers because both species are known to displace native species and modify habitats at high densities. Various features of a potential reserve site are essential for the reserve to be effective, such as the ability of the area to retain larvae of target-protected species, but this key concept can fail to preserve the areas native biota if non-indigenous species are taking advantage of this. Another study within the same reserves revealed that a non-indigenous clam species *Venerupis philippinarum* was highly abundant in marine reserves compared to outside the reserves (Byers 2005). A native clam species *Protothaca staminea* is ecologically and morphologically similar to *Venerupis* yet shows no differences in abundance within the reserves. Both species are heavily harvested yet *Venerupis* substantially benefits from the reserves due to its shallower burial depth. No scientific data exists on whether non-native marine species will be similarly enhanced in New Zealand marine reserves.

Prediction and prevention of non-indigenous species

Due to the damaging economic and ecological costs that invasive species impose there has been an urgent need for the recognition and prevention of invasions before they occur. Early research focused on the damage caused after an invading species had already established, but this proved to be inefficient as the eradication of such species was nearly impossible. Research on the invasion of species is now treated as a sequence of events instead of a single episode. The invasion process can be broadly categorised into five sequences: 1-Entrainment which involves uptake of the organism (for example, larvae taken into ballast water), 2-Transportation, 3-Introduction, 4-Establishment, and 5-Spread (Kolar & Lodge, 2001). Studying each transition separately is important to fully understand the characteristics of species important for successfully establishing in a region (Kolar & Lodge, 2001). Chapman and Carlton (1991) have developed a framework of ecological, evolutionary and geographical characteristics as predictive criteria for the identification of invasive species. A criterion for being an invasive species is to be predominantly associated with or restricted to human created structures such as pilings, jetties, floats and boat-bottoms (Chapman & Carlton, 1991).

Specific aims for research

The aims of this research are to determine whether 1) marine reserves enhance traditional species within intertidal environments, 2) marine reserves enhance non-indigenous species, 3) constructed habitats in the intertidal area will also provide native species traditionally used by Māori with a suitable habitat, and 4) constructed habitats enhance non-native species populations in New Zealand.

An alternative hypothesis is that constructed environments will attract non-indigenous species, which will lead to declines in native species. If we discover that this is occurring then we can inform management authorities so that they can re-evaluate how to manage these habitats.

METHODS

Marine Reserves

Study sites

Two marine reserves and one marine park were investigated for differences in species abundance inside and outside of each protected area; Cape Rodney-Okakari Point marine reserve, Te Tapuwae o Rongokako and Mimiwhangata Marine Park.

Cape Rodney-Okakari Point marine reserve was established in 1975 near Leigh in the northern North Island, New Zealand (36^{o} 17'0"S/174^{o} 49'0"E). This is the oldest marine reserve where it protects 547 ha of marine life, and has unique oceanographic features due to the protection that Goat Island provides. This reserve has coarse sandy beaches and intertidal rocky platforms that are inhabited with diverse communities of marine life (Enderby & Enderby, 2006). I sampled this reserve on January 5, 2009 and sampled Matheson Bay (reference beach) on January 6, 2009, at approximately the same time of day. My reference beach was restricted to one side of the reserve because it had relatively comparable habitats with similar wave exposure.

Te Tapuwae o Rongokako is located on the east coast of the North Island in Gisborne, New Zealand (38^{o} 39'11"S/178^{o} 0'15"E). The marine reserve was established in 1999 and protects an area of 2452 ha. This coastline is semi-exposed with intertidal sandstone reefs and sandy beaches (Enderby & Enderby, 2006). I sampled this reserve on March 4 2009 and Makorori Beach (reference site) on March 5 2009, at approximately the same time of day.

Mimiwhangata Marine Park was established in 1984 and is located just north of Whangarei in northern New Zealand (35° 25’60”S/174° 25’0”E). This marine park protects 2410 ha and consists of sandy beaches with intertidal rocky reefs. Black nerita, cat’s eyes, and oyster borers are abundant within intertidal regions, while Tuatua and morning star shells are common within the sand flats (Enderby & Enderby, 2006). I sampled this reserve on August 4 2009 and Oakura beach (reference site) on August 5 2009 at a similar time of the day. The sampling opportunities were limited in these studies due to sampling having to take place during low tides.

Reserve selection and target species

Reserve sites were selected based on both the presence of cat’s eyes *Turbo smaragdus* and kina *Evechinus chloroticus* as well as having a rocky intertidal area. These features reflect traditional harvesting grounds and they are preferred habitats for the species that were studied. Limitation of suitable rocky platform beaches meant that only three marine protected areas were used in this research. Initial reconnaissance surveys for all sites were required to ensure that each reserve had similar topography and wave action features to its corresponding reference beach.

Field collections

During each reconnaissance survey, spot checks were used to examine the target species to ensure that species abundance was sufficient for this study. Five areas (each 10 m^2) inside each reserve were randomly selected for sampling and the same process was used for each reference site. Each area (10 m^2) was divided up into a grid so that each metre interval along the x-axis and y-axis could be used for co-ordinates. Two randomly generated numbers would provide the co-ordinates that would indicate the position for sampling to take place on the grid. The abundance of *Turbo smaragdus* was sampled using a plastic quadrat (25 cm^2). A total of ten quadrats within each $10m^2$ were used to sample the abundance of *Turbo smaragdus*. The plastic quadrat was created using ordinary plastic garden stakes; this material was found to be more appropriate over a wooden quadrat as they dry faster and are lightweight.

Within each of the five 10 m^2 study areas, inside and outside each reserve, all kina (*Evechinus chloroticus)* were counted within the area. The behaviour of this species is often to take shelter within small cracks or under ledges and then use its spines to cause abrasion to the surrounding rock to allow for more room. They were sampled using different methods because their abundances differed

Two known non-indigenous species *Balanus trigonus* and *Hymenium percleve* were to be studied inside two of the reserves. *Balanus trigonus* is an invasive barnacle, which is found under rocks, around the low tide mark and in the subtidal regions of the Whangarei and Leigh shore (Foster, 1967). *Hymenium percleve* is a common intertidal sponge in New Zealand and both were thought to be good candidates for this study. However, during the reconnaissance survey the abundance of both species were not sufficient within each reserve to be sampled and statistically quantified.

Data analysis

Mann-Whitney *U* tests were used to compare the abundances of each target species inside and outside of each marine protected area. This analysis is a non-parametric test, and was used to analyse the data due to both unequal variances between reserve sites and their corresponding references sites, even following transformation, and the presence of zero values for some species at some sites. Mann-Whitney *U* tests were undertaken using STATISTICA (version 8.0. StatSoft, Inc, 2007).

Constructed Artificial Habitats

Study sites

A total of 14 different sites were sampled around the North Island where selection of each site was dependent on the presence of artificial structures such as wharves and bridge pilings and comparable nearby natural rocky reefs that were usually within 1 km. My study sites were on both the west and east coasts of the North Island and included Gisborne, Tolaga Bay, Tauranga, Whakatane, Devonport, Bayswater, Birkenhead, Whanagarei, Coromandel, Raglan, Port Waikato, Kawhia, Mahia and Manukau harbour (Fig 1). Sampling for this section took place from January through to August 2009.

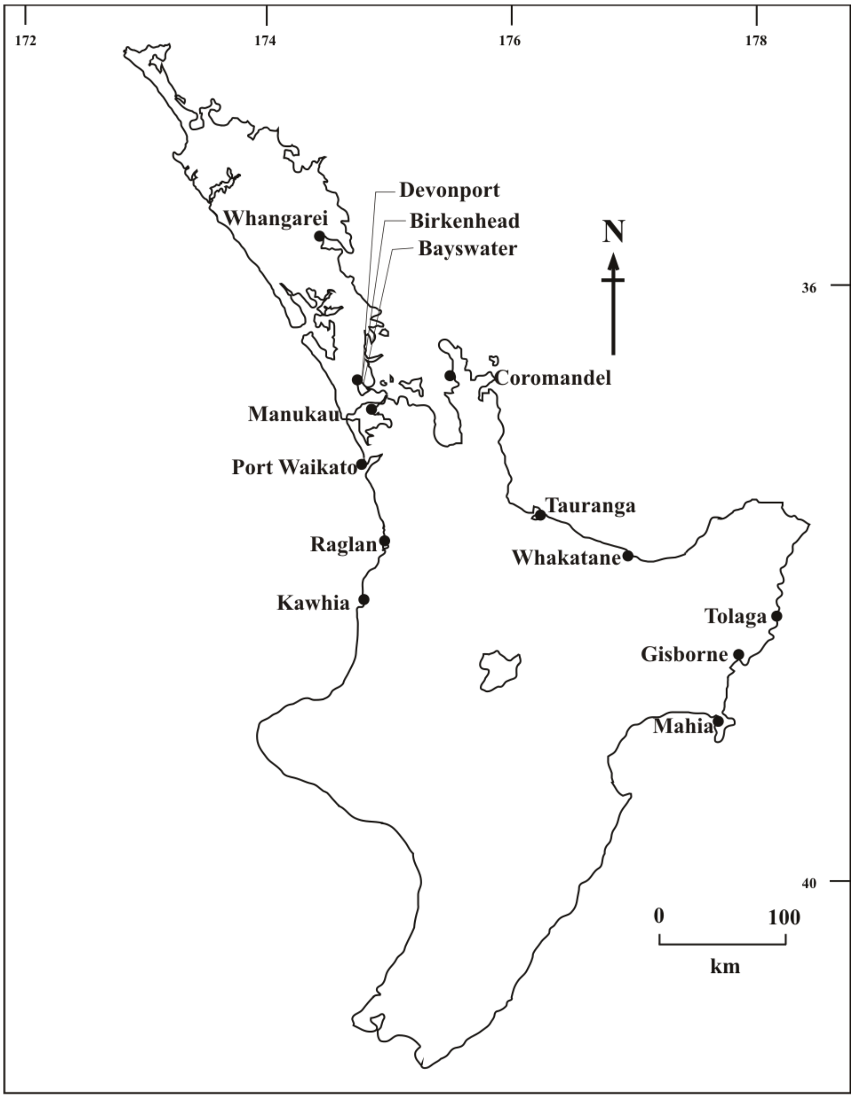

Fig 1. Map of North Island, New Zealand showing distribution of constructed sites.

Field collections

This study compared organisms living on “artificial” structures with those living on nearby natural rocky areas within the intertidal region. Using a 15 cm^2 quadrat (similar to the one used in the previous method) and a putty or diving knife, I scraped all organisms that were within the area into a container during low tide. This was replicated three times on each substrate, on different sides and heights of the same piling, for example, to obtain a good representation of the community composition. These replicates were combined into a single sample for each site for analysis. Depending on what substrates were available, I sampled wooden and concrete habitats and nearby rocky areas. Each sample was preserved in 95% ethanol and later identified in the laboratory (using ID guides from the following references Bucknill &Powell, 1924; Foster, 1967; Walsby *et al.* 1982; Morley & Anderson, 2004). A total of 93 samples were collected from concrete, wooden and rock substrates.

Data analysis

Multi-dimensional scaling (MDS) was used to explore trends in the community composition among sites, and assist in making assumptions as to which habitat types are more associated with non-indigenous and kaimoana species. The data was transformed, Log (x+10), prior to analysis to down weigh the influence of abundant species. This method is appropriate for this data set because it has a non-normal distribution and it provides visual results that can be easily interpreted. MDS creates a 2-D map of samples to demonstrate how similar they are depending on their proximity to one another. A stress value on the map is used to measure the goodness of fit where zero indicates a perfect fit. The stress value calculates how well represented the distances are on the MDS map with the original distance matrix. For this study the PRIMER 4.0 statistical package was used to carry out an MDS analysis; this was performed on a ranked similarity matrix that was utilised based on the Bray-Curtis similarity co-efficient (Clarke & Warwick 1994).

Analysis of similarities (ANOSIM) is a non-parametric permutation test that is applied to the multivariate data (Clarke & Warwick 1994) to detect similarities in community assemblages for the different sample sites. This analysis was applied to the same similarity matrix as above, to test for significant differences between the sample community assemblages and the different habitat types (rock, concrete, wood). ANOSIM is similar to the MDS analysis in that it also measures the dissimilarity of samples, but instead of displaying the results as a map, ANOSIM produces an *R*-statistic where its value is between 0 and 1. Samples that are close to zero suggest that groups are very similar to one another whereas samples that approach one imply that these groups are dissimilar. The PRIMER 4.0 statistical package was used to carry out an ANOSIM test for the relationship between species in a community and their corresponding habitat type. ANOSIM was performed on the Bray-Curtis dissimilarity matrices and 999 permutations were performed.

A shade diagram was constructed by re-sorting the original data matrix in order to indicate which species were associated with the rock, wood and concrete substrates. The samples were ordered on the x-axis by substrate and sites. The order of each site was based on geographical location, arranged from east coast sites along to west coast sites. On the y-axis, species were ordered by cluster analysis (Primer 4.0, Plymouth Marine Laboratory) of samples based on taxonomic composition (Bray-Curtis similarity, fourth-root transformed abundance data). On the re-ordered matrix, the increasing abundance (fourth-root transformed) of each species is plotted as a greyscale using Spyglass Transform V.3.3.0 (Fortner Research LLC), a computer program that produces greyscale plots from 3-D arrays of numbers. Shade diagrams are useful for visualizing the relationships of species to sample sites and habitat preferences because large-scale patterns can be assessed.

Because these statistical formulas are intended to calculate differences or similarities between community assemblages, some sites or habitats were excluded (Manukau, Mahia, Coromandel, Whangarei) as their assemblages contained only single species.

Fauna Associated With Green-Lipped Mussels

Green-lipped mussels (*Perna canaliculus*) were bought from ten different supermarket stores, and the fouling organisms attached to their shells were identified and enumerated. Approximately one kilogram of mussels was bought from ten supermarkets: four different supermarkets in Hamilton, four in Auckland, one in Gisborne and one supermarket in Whangarei (Fig 2).

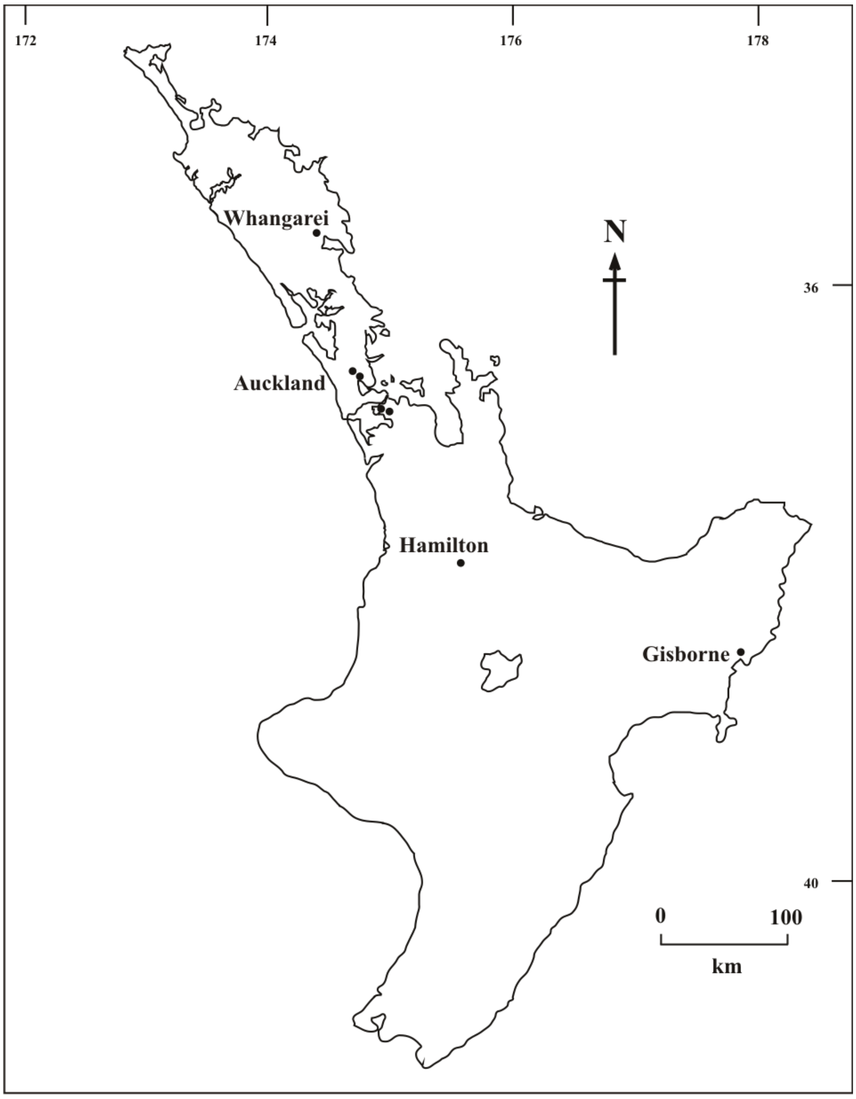

Fig 2. Map of North Island, New Zealand showing distribution of towns where green-lipped mussels were bought from various supermarkets.

RESULTS

MARINE RESERVES

Cape Rodney-Okakari Point (Leigh) reserve

Mann-Whitney U comparisons inferred that kina abundance inside the marine reserve was significantly higher compared to outside the reserve ($p<0.05$) (Table 1). The average abundance for kina inside the reserve was 127 compared to the relatively low average of 37 kina found outside in unprotected areas (Table 2). There were no significant differences found between the reserve and the reference site for Cat's eyes ($p>0.05$).

Table 1. Results (*p*-value) from Mann-Whitney U tests comparing species inside reserves compared to unprotected beaches. **p*-values are significant (a=0.05)

	Leigh	Mimiwhangata	Gisborne
Kina	0.028*	0.917	0.009*
Cat's eyes	0.917	0.076	0.037*

Te Tapuwae o Rongokako (Gisborne) reserve

Mann-Whitney U tests revealed a significant difference for kina abundance inside the reserve compared to outside the reserve ($p<0.01$). The average abundance for kina inside the reserve is 14, which was significantly higher compared to the low 0.2 average for kina at reference beaches. Cat's eyes density was significantly lower inside the reserve compared to outside at the reference beach ($p< 0.05$). The average abundance of cat’s eyes inside the reserve was only 62 compared to 143 cat’s eyes found at the reference beach (Table 2).

Mimiwhangata (Whangarei) Park

Abundances of both species were not found to be different compared to the communities living outside the reserve using Mann-Whitney U tests ($p>0.05$). Average kina abundance in Mimiwhangata was 2, and kina abundance was 3 in the unprotected reference beach. Average abundances of cat’s eyes were 228 inside Mimiwhangata and 141 in the unprotected beach.

Table 2. Average abundances of kina and cat’s eyes in reserves and the unprotected beaches

	Gisborne Reserve	Makorori	Mimiwhangata Reserve	Whangarei	Leigh Reserve	Matheson Bay
Kina	14	0.2	2	3	127	37
Cat's eyes	62	143	228	141	63	73

CONSTRUCTED STRUCTURES

Habitat associations

Despite sample sites being spread around the North Island, community assemblages on artificial habitats were generally relatively similar to one another, as indicated by the MDS ordination. Wood and concrete habitats mostly clustered closely together in the top right hand corner of the ordination (Fig 3). ANOSIM confirms the similarity between artificial habitats, with the low *R-value* (-0.028) calculated between concrete and wood habitats ($p>0.05$; Table 3). Natural rocky substrates had fairly diverse communities among sites as they are spread across the MDS plot. However, ANOSIM indicated significant differences between natural rocky areas and wooden substrates ($R=0.269$; $p<0.05$) but not between rock and concrete substrates (0.057; P>0.05). Birkenhead samples were the major exception to the samples grouping by habitat, and likely affected these results. This anomaly may be explained as the ‘wood habitat’ sampled there was a tree that was situated in the high intertidal (and therefore natural), and the concrete habitat was steps leading directly onto natural rocky reefs. These artificial habitats are associated around the natural habitats on the MDS as they had the same native species associated with them. Tolaga Bay concrete samples also separated off from the main cluster of concrete and wood samples; the rock and concrete habitats had similar species composition in this locality.

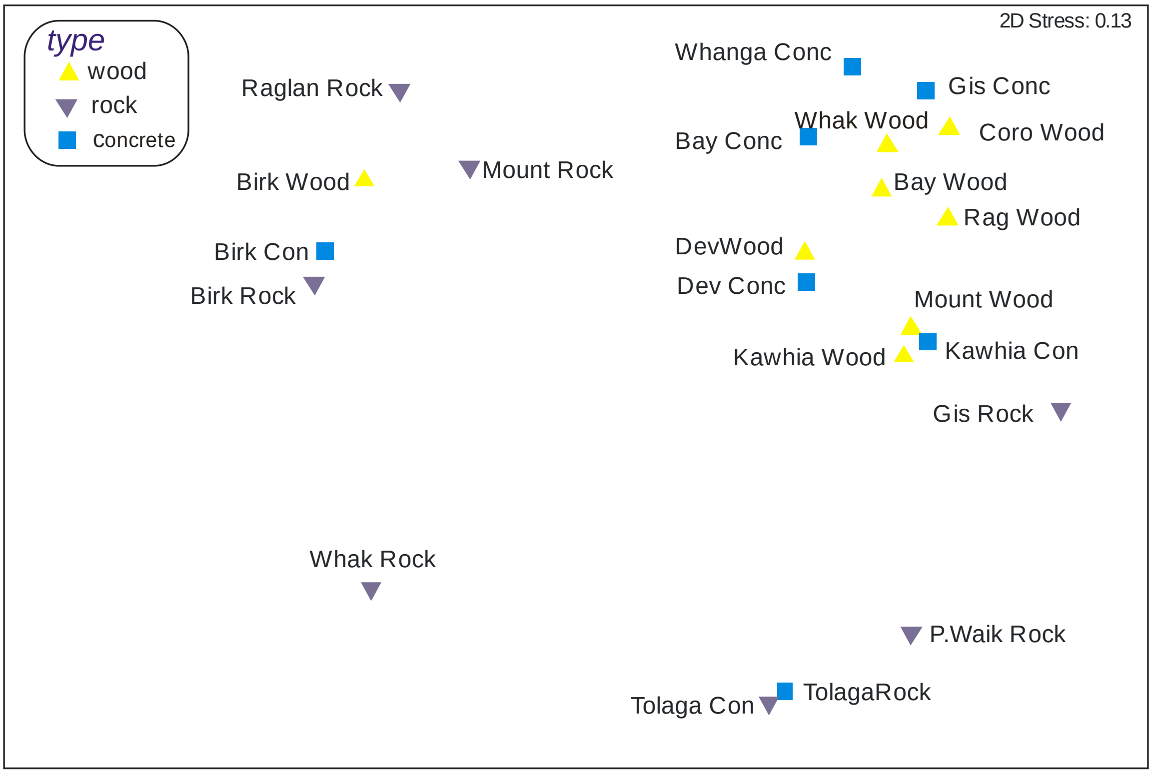

Fig. 3 Multidimensional scaling (MDS) plot of samples based on communities associated with wooden, concrete and rock habitats. (Birk=Birkenhead, Whak=Whakatane, P.Waik=Port Waikato, Mount=the Mount in Tauranga, Bay=Bayswater, Whang=Whangarei, Dev=Devonport, Rag=Raglan).

Table 3. Results (*R*-values) from ANOSIM tests for habitat related differences (Global *R*=0.1) **p*-values are significant (a=0.05)

	Wood	Rock	Concrete
Wood	~	0.269*	-0.028
Rock		~	0.057
Concrete			~

Species-specific associations

Artificial habitats had fairly similar community assemblages that were dominated by the barnacle *Elminius modestus* and the black mussel *Xenostrobus* (Fig 4). Pacific oysters were common in concrete habitats and were relatively rare in the natural habitats. The native rock oyster, on the other hand, was found at approximately the same relative frequency in natural and constructed habitats. While *Elminius modestus* was the dominant barnacle associated with artificial habitats, it was found relatively infrequently in natural habitats. *Elminius plicatus* and *Chamaesipho columna* were more common in the natural rocky reef habitats than in constructed habitats. The small barnacles and mussels dominated the samples by abundance but oysters dominated each sample by their area covered and size.

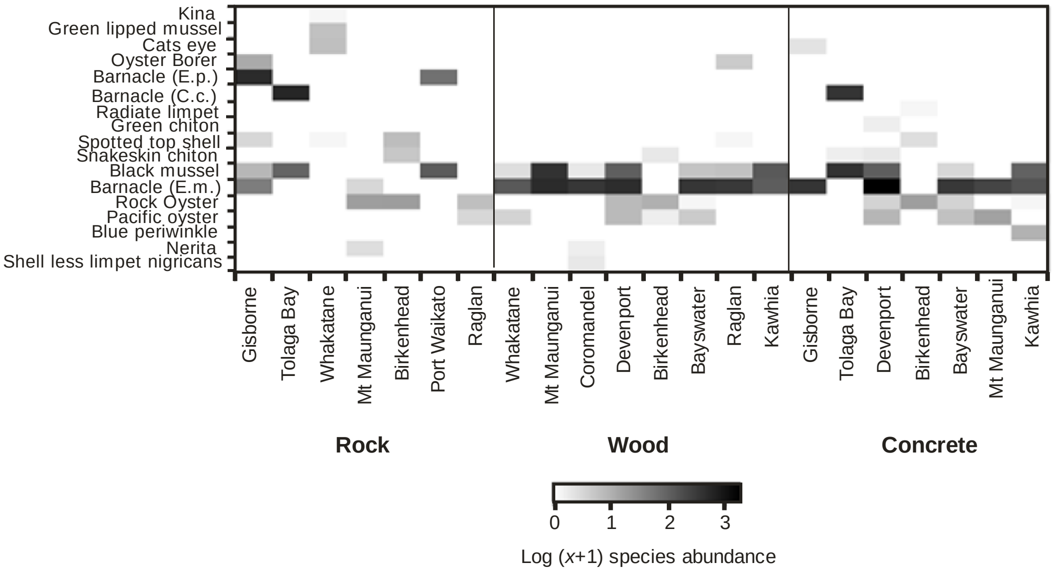

Fig. 4 Shade diagram for species and study sites divided into habitats. Darker boxes represent a greater abundance of species for that particular habitat. Rows and columns are ordered based on study sites along North Island coastline starting from Gisborne to Kawhia. Sample sites with only single species were not used for this analysis (Mahia and Manukau). Barnacle abbreviations are: E.m=*Elminius modestus*, E.p= *Elminius plicatus*, C.c= *Chamaesipho columna*

There were a total of 13 species found on rock habitats, 11 total species were recorded on concrete habitats and 9 total species recorded on wooden habitats. There were 8 different wooden sites that were studied with a total of 24 samples taken on this substrate (3 samples per site). There were 7 different rock sites that were studied with a total of 21 samples taken on this substrate. This also applied to concrete habitats where there were 7 sites and a total of 21 samples collected. Even though there were 8 sites for wooden substrates and only 7 for concrete and rock habitats, there was no significant difference as the average species richness was 3 species per substrate. Even though species richness was relatively similar between substrates there were certain species that were exclusive to each of the habitats (Fig 4). Species found only on concrete habitats include the small blue periwinkle (*Austrolittorina antipoda*), the green chiton (*Amaurochiton glaucus*) and the radiate limpet (*Cellana radians*). The shell-less limpet (*Onchidella* *nigricans*) was found only on wooden habitats in relatively low abundance. The green-lipped mussel (*Perna canaliculus*), kina (*Evechinus chloroticus*) and cat's eye (*Turbo smaragdus*) were all recorded in Whakatane and are only recorded on rock habitats.

Fauna associated with green-lipped mussels bought from supermarkets

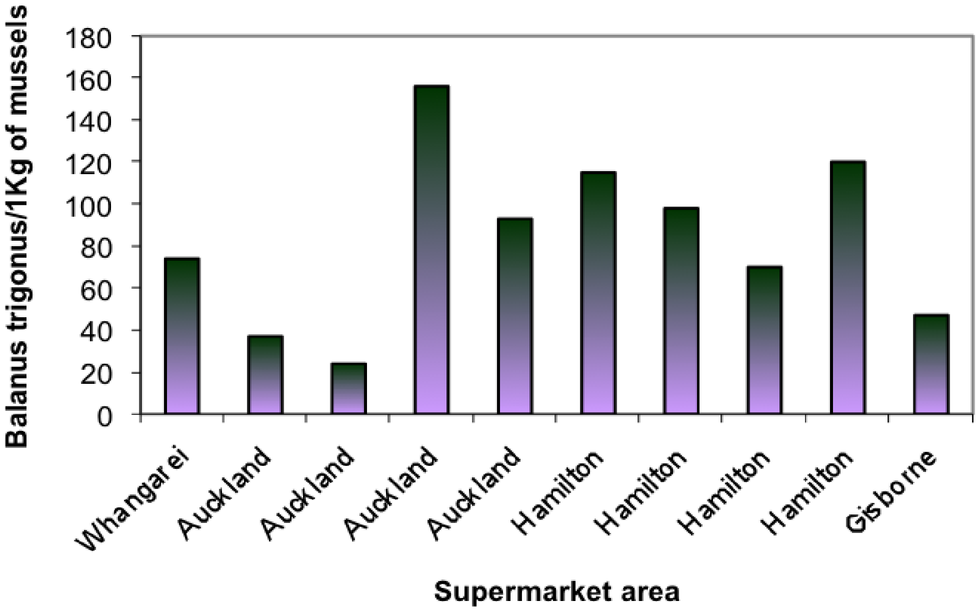

Fig. 5 Graph representing the total abundance of *Balanus trigonus* associated with green-lipped mussel shells (1kg per supermarket).

The abundance of *Balanus trigonus* associated on mussel shells is relatively variable around the North Island supermarkets. The highest and lowest figures were recorded in specimens collected in the Auckland supermarkets and barnacles ranged from 24-156 per kilo of mussels. The average barnacle abundance recorded was 83 across these North Island supermarkets.

The white slipper shell (*Crepidula monoxyla*) is native to New Zealand and although it was associated with green-lipped mussels it is not harvested for food. This species was recorded at relatively lower abundances compared to *Balanus trigonus* and had an average abundance at 3 specimens across these supermarkets.

DISCUSSION

The knowledge gained from my research is primarily intended to provide the Māori people of New Zealand information about the availability, enhancement, and/or protection that marine reserves, marine parks and artificial habitats offer intertidal kaimoana. While the intentions of many marine reserves in New Zealand primarily are to sustain fishery species, there has been a shift to establish reserves to protect and maintain our unique marine biodiversity (Enderby & Enderby, 2006).

MARINE RESERVES

The abundances of kaimoana inside and outside marine reserves

Kina (Evechinus chloroticus)

Kina were significantly higher in reserves relative to reference sites at Leigh and Gisborne, but differences were not significant at Mimiwhangata. The kina abundance data collected from the Leigh marine reserve indicate that the protection is beneficial for this species. This species is abundant throughout the subtidal regions around New Zealand but the ecology of this species in regulating communities is not well known for intertidal regions (Andrew, 1988). Numerically kina density was relatively low inside Gisborne reserve, but because there were few kina found outside the reserve, statistical tests infer that this difference is significant. Similar density studies on intertidal kina in Gisborne have shown an opposite trend where kina density tended to be higher outside the reserve at reference beaches (Freeman, 2006). It is possible that differences in sampling duration could result in these differences as Freeman (2006) sampled similar regions over a period of five years and could obtain accurate trends in population dynamics, whereas my sampling was a single event in 2009. An alternative trend could be that since Freeman's (2006) sampling regime, which was conducted up to 2004, there may be a slight recovery of kina, which has allowed them to replenish within the Gisborne reserve.

Abundance statistics suggest that there were no significant differences between the kina populations living inside the Mimiwhangata Marine Park compared with those living outside. This is a marine park where commercial fishing is prohibited but limited recreational harvesting is allowed for traditional purposes.

Ataata/ Pupu (Turbo smaragdus)

In the Gisborne marine reserve the abundance of cat's eyes were substantially lower in comparison to the populations of cat's eyes living outside of the reserve. The cat's eyes abundance data collected from the Leigh marine reserve and Mimiwhangata Marine Park indicates that there was no significant difference between the populations living inside compared to outside references beaches. The findings from the Gisborne reserve are opposite to what I expected to find as this traditionally harvested food source shows no significant response to the protection that reserves offer. This turbinid gastropod is abundant around New Zealand's intertidal and estuarine habitats and reaches up to 50 mm in width (Alfaro et al. 2007).

Factors affecting kina abundance in reserves compared to unprotected sites

Harvest restriction

This study concentrated on intertidal-shallow subtidal (less than 1 metre) habitats where kina preferentially took shelter within crevices and under ledges. This species takes shelter within crevices generally at sizes up to 20-50 mm, and then leave to live in an open and less secure environment (Andrew, 1988). Kina living in these habitats may be significantly more abundant inside the Leigh reserve purely due to having 35 years protection from being harvested by humans. This protection will promote reproducing individuals to successfully recruit offspring to sheltered crevices such as those in this study.

A considerable Māori community is present in Gisborne and gathering of seafood is almost a weekly ritual along the coastline. As this species is considered a delicacy, seafood gathering is likely a dominant factor that would be regulating seafood populations outside the protected areas (Gibson, 2005). Differential recruitment is a possibility that can enhance kina populations inside the reserve. Intertidal populations are entirely dependent on populations living in subtidal regions (Freeman, 2006) and it is possible that a higher proportion of adult spawning kina inside the reserve are contributing to the number of intertidal kina.

In the Mimiwhangata Park there was no significant difference found for kina or cat’s eyes compared with the neighbouring coastline, and this fact may be attributed to the limited protection offered by the marine park. Mimiwhangata Marine Park is a special kaimoana harvesting ground for Māori people and there are pa and garden sites that are still visible as evidence of long-term Māori residence (Enderby & Enderby, 2006). One crayfish pot per person is permitted and this would have a definite impact on the sustainability of the species. The removal of dominant predators such as crayfish and snapper from the ecosystem will allow kina to inhabit subtidal regions and be free to graze on kelp forests. This may promote the shift from intertidal dwelling kina to move out to subtidal regions where the area is free from predators. These factors may distort my results into reporting an insignificant difference of kina with reserve protection, when in fact the kina population maybe highly abundant in the park within subtidal regions. This theory is supported as kina has been observed to be highly abundant within the marine park, where it dominates the subtidal regions up to 6 m deep, and has left barrens due to overgrazing on kelp forests (Enderby & Enderby, 2006).

Denny and Babcock (2004) have demonstrated that snapper species (*Pagrus auratus*) have been unable to recover and increase in abundance within Mimiwhangata marine park, yet this species has dramatically increased in total-ban reserves such as the Poor Knights and Leigh (Shears *et al.*, 2006). Shears *et al.* (2006) have suggested that limited protection in Mimiwhangata Marine Park does little to protect targeted species, and recreational fishermen maybe the cause. Kina, green-lipped mussels, crayfish, scallops and tuatua can all be harvested within this marine park (DOC, 2007). Studies of long term monitoring of the crayfish *Jasus edwardsii* showed no response to the restricted protection, and a decline in the population that could be attributed to the regional catch per unit effort estimates (Shears *et al.* 2006).

Evidence of differential harvest pressure has been demonstrated by Shears *et al.* (2006) who examined no-take reserves versus partially protected reserves. While urchin barrens have dominated the area since the 1970s in Mimiwhangata, there is a shift in predatory dominance in the nearby Tawharanui Marine Park that is exclusively a no-take zone. A trophic cascade is evident as snapper and crayfish are now dominant and can control kina abundance thereby allowing the re-establishment of kelp forests.

Predation

Factors regulating kina abundance include predators, food supply, and successful recruitment to crevices (Andrew, 1988). Known predators of kina include the crayfish *Jasus edwardsii*, snapper *Pagrus auratus*, the whelk *Charonia capax*, and the starfish *Coscinasterias muricata* (Andrew, 1988). Leigh and Tawharanui have higher predator densities inside the reserves compared to outside areas; snapper is 5.8-8.7 times more abundant inside these reserves and crayfish are 1.6-3.7 times more abundant (Babcock *et al*, 1999). Snapper and crayfish are both known to predate on kina inside reserves but these maybe restricted to kina living in the open areas and not kina living in crevices in the intertidal. The predatory aspects on kina were not part of this investigation, but I did observe that there was a high abundance of starfish at the Leigh reference beach and many were in the process of digesting kina. I suggest that it is the starfish that are regulating kina populations outside of the marine reserve; Shears & Babcock (2002) have also made observations that outside of the reserve the starfish (*Coscinasterias*) and whelk (*Charonia*) were dominant predators regulating kina. They recorded both species to occur at lower densities inside reserves compared to non-reserve sites. Starfish feed by everting their stomach outside of the body and digest their prey that way before retracting the stomach back inside of its body (Anderson 1954; Mauzey, 1966). This may contribute to the lower kina population abundance found outside the reserve because starfish seen in the reserve were typically seen only under boulders or in rock pools. Snapper and crayfish are abundant inside reserves and can predate on kina living in subtidal habitats, but this maybe restricted to these regions because crevice habitats in the intertidal maybe relatively inaccessible. This could be the reason for why I found kina at high densities inside Leigh. However, the starfish and whelk predators can access kina at reference sites because they can fit into the same spaces. These differences in predators can help explain why kina are abundant inside Leigh.

Freeman (2008) has conducted extensive tagging surveys on crayfish *Jasus edwardsii* in the Gisborne marine reserve and the population has recovered in the 11 years of protection. Kelly *et al*. (2000) have studied the recovery rate of this crayfish in several North Island reserves and have found that there is an increase in population density of 3.9% in depths less than 10 m. Crayfish have been observed to migrate within intertidal reefs in the Gisborne reserve during high tides to forage for food; a behaviour that had not been observed by the species previously (Freeman, 2008). In my study of the Gisborne reserve kina were found more abundant inside compared to the reference site. This may provide evidence the crevice habitats provide shelter in the intertidal, especially since crayfish forage in this region during high tide.

Denny and Babcock (1994) suggest that partially protected reserves, such as the Mimiwhangata Marine Park, are unsuccessful for conservation measures, as reef fish such as snapper showed no difference in size and abundance compared with heavily fished areas. Being labeled a marine park gives the misconception that the area is abundant with abundant marine life, and so the opposite effect is achieved where fishing is concentrated in these areas (Denny and Babcock, 1994; Shears *et al.*, 2006). Deleterious effects have been observed in marine reserves and nonprotected coastlines from a variety of human impacts that lead to unnatural disturbance regimes. The Department of Conservation compiled a survey that assesses the effects that human impacts have on New Zealand's coastline and marine protected areas (McCrone, 2001). Trampling, harvesting, diving, and boating are found to have negative impacts on New Zealand's coastline. The obvious impact being destruction of marine life and the second major impact is the alteration of community composition via selective removal of species during harvesting.

Factors affecting Ataata abundance in reserves compared to unprotected sites

Harvest restrictions

One major factor that is important for interpreting the lower numbers inside the Gisborne marine reserve, and insignificant differences elsewhere, is the preferential collection of seafood. Reserves are often established due to over exploitation of target species, but there may be differences in what kaimoana type is preferred in different localities. To better interpret these results it would require a survey on preferred kaimoana collected. There are records for important intertidal kaimoana for Gisborne and central Hawkes Bay regions (Gibson, 2005; personal observation), but important kaimoana species are not generally known for the Leigh and Whangarei regions. Cat's eyes are traditionally important species for Māori people in the Gisborne region where they are still harvested for consumption today (Gibson, 2005; personal observation). However, preferential harvesting of this species in the Gisborne region does not explain the differential patterns observed, because cat's eyes were significantly higher outside the reserve where harvesting occurs.

Competition and predation

There are many possible ecological mechanisms regulating and restricting the population of cat's eyes species inside the Leigh reserve. One possible reason could be enhanced intraspecific and interspecific competition for limiting resources. Food sources such as algae found in the higher shore elevations are often consumed at a faster rate compared to algae living at the low shore elevations. This is a consequence of many grazing gastropods preferentially inhabiting higher shore elevations in order to escape predation (Fawcett, 1984). However, cat's eyes are mobile grazers that are affected by tidal inundation and therefore migrate upward during high tides and then shoreward as the tide recedes (Alfaro, 2006). This daily migration is driven by feeding activities as it allows exploitation of a greater food supply (Alfaro, 2006). Interspecific differences in grazing pressure can possibly restrict cat's eyes from increasing in abundance within the reserve. Fawcett (1984) observed that the abundant macro-algae found on the lower intertidal shore could not be controlled by grazing gastropods, but were controlled by sea urchins. The effect that urchins have on macro-algae could possibly be exacerbated, as the reserve has enhanced populations of kina and this may reduce the available macro-algae for cat's eyes.

Known predators of cat's eyes are two predatory snails that include *Haustrum haustorium*, and *Dicathais orbita* and starfish such as *Astrostole scabra* (Alfaro, 2006). Cat's eyes may risk this potential threat posed by its predators, and migrate shoreward during low tide, as they have possession of a heavy operculum.

Bullock (1953) found that the gastropod *Acanthina spirata* did not respond to threats posed by a predatory starfish, and he concluded that this was also due to its thick operculum.

Therefore, the possibility that kina are superior competitors for food inside the reserve, which would restrict cat's eyes from becoming abundant, is compelling. Abilities of kina to outcompete cat's eyes for food inside the reserve would be negligible outside the reserve because I observed high densities of starfish at Matheson's Bay. Starfish are known to prey on both cat's eyes and kina; Duggins (1983) observed that starfish, which resided in the intertidal, would mainly feed on gastropods, whereas starfish that lived in subtidal regions fed on urchins. Bullock (1953) suggests that when gastropods and predatory starfish both occur, there is a chemical signal specific to predatory starfish that will keep gastropods at a safe distance. In his work Bullock was able to demonstrate that there was an obvious flight response from gastropods when touched by a starfish tube foot. Duggins (1983) also observed a flight response in reaction to predatory starfish. For my research kina were typically counted inside cracks and crevices, and this would definitely restrict their movements (Bullock, 1953; Duggins, 1983). Starfish which feed on both cat's eyes and kina may find that searching time and handling time for their prey to be much more efficient on kina, compared to cat's eyes, since cat's eyes species are not restricted to the cracks and crevices. Kina may compete better for food inside the reserve and this would explain the lower abundance of cat's eyes in relation to outside the reserve. But because of the abundant starfish found outside of the reserve, kina may find it more difficult to feed if it were heavily predated upon, and this would allow cat's eyes to feed more efficiently since they are not restricted to cracks and have a flight response toward starfish (Duggins, 1983).

The direct and indirect effects of trampling on rocky shores

The effect of trampling could also be responsible for the low abundance of Cat's eyes inside the Gisborne marine reserve relative to outside. Along the south-eastern rocky shores of Australia, for example, Povey & Keough (1991) found that the gastropod *Turbo undulatus* was reduced in abundance in trampling experiments. *Turbo undulatus* feeds on the most dominant algae on the south-eastern Australian shores, which is *Hormosira banksii* (Povey & Keough, 1991). This algae is also a dominant plant in New Zealand that *Turbo smaragdus* is known to feed on (Schiel & Taylor, 1999). *Turbo smaragdus* has also been observed grazing within the encrusting and turfing coralline algae, which make up the understory region beneath *Hormosira banksii* in New Zealand (Schiel & Taylor, 1999). Trampling across rocky shores has an obvious destructive effect on marine animals and is known to have a more significant effect on hard-bodied gastropods than soft-bodied anemones, because trampling over algae removes the food sources for these herbivorous gastropods (Brown & Taylor, 1999).

The effects that foot traffic has on intertidal marine organisms may also be a contributing factor as to why we are not seeing a significant increase in cat's eyes inside Leigh reserve. This species feeds out in the open reef platforms and may therefore be susceptible to marine reserve visitors. Schiel and Taylor (1999) found that the trampling effects could reduce areas that had >96% canopy coverage of *Hormosira banksii* down to 25% after a single tide. They concluded that higher trampling densities caused considerable bare space and could take up to a year for new recruits to become established and help recover damaged areas. Although disturbance plays an important role in near-shore dynamics, the foot traffic within the reserve may contribute to the relatively low population abundance of cat's eyes. Leigh marine reserve is an important restoration environment where many scientists and visitors are attracted to study marine life; and while its popularity is beneficial in raising awareness the unintentional stepping on marine life is a consequence.

Constructed Artificial Habitats

Habitat associations

For this part of my research I aimed to determine if a) artificial habitats were utilised by native kaimoana, or if b) they provided suitable habitats that encouraged the settlement of non-indigenous species. The Multidimensional Scaling plot indicated that species that occurred on man-made habitats tended to cluster together, indicating there was generally not a great variety in the composition of fauna on constructed habitats. These samples were collected in different localities around the North Island, which suggests that regardless of whether sites are located on the east or west coasts of New Zealand, the community composition of these habitats are relatively similar. The communities found on natural rocky shores, in comparison, were more spread across the Multidimensional Scaling plot, suggesting a higher level of diversity among areas.

Dominant species on constructed substrates

The barnacle *Elminius modestus* was commonly the most abundant species found in constructed habitats. This species has previously been reported as an early successional species on wharf piles where it is tolerant of low salinity and turbid waters (Morton and Miller, 1968). Important for its dominance in constructed habitats, *E. modestus* is commonly known to dominate the rocky intertidal regions around North Island’s coastline and has a wide vertical range; in sheltered cracks it can extend well above Mean High Water Neap (MWHN) and in suitable places can extend down to Mean Low Water Neap (MLWN) (Morton and Miller, 1968).

As this species is one of the most common barnacles in natural habitats, it therefore has ample opportunities for young to colonise constructed habitats. The construction of wood and concrete structures in areas with soft shores, where this species is less likely to do well, provides hard surfaces that allow the settlement of planktonic larvae. For example, Whangarei, Kawhia and Raglan are all examples of areas surrounded by soft shores where this barnacle is well established on concrete and/or wooden structures. Recruitment and settlement are critical for the survival of sessile species, particularly since relocation after settlement is not possible (Raimondi, 1988). High fecundity is therefore another positive trait that *E. modestus* possesses, which gives it competitive advantage over other barnacle species that compete for space. The breeding season for *E. modestus* is continuous throughout the year, and juveniles take only two to three months to grow to maturity (Moore, 1944). Finally, the possession and tolerance of its planktonic larval stage of this species allows for dispersal to constructed habitats, providing another reason for their success. The larval stages of *E. modestus* are eurythermal and euryhaline, defining characteristics that permit them to survive in a wide range of habitats and are suggested to be the reasons for their spread around European waters (Harms, 1999). In contrast, the predatory gastropod *Lepsiella scobina* is known to be relatively scarce on wharf piles because it has no free-swimming larvae and therefore distribution to wharf piles is difficult (Morton and Miller, 1968). This predatory gastropod was present at Raglan wooden habitats and in relatively low abundances.

The little black mussel or flea mussel *Xenostrobus pulex* was also a dominant component of constructed structures. *Xenostrobus pulex* is native to New Zealand waters where it dominates fouling communities and forms dense covers in the spring (Menzel, 1991). Morton and Miller (1968), for example, observed the mussel dominating wharf piles above the mid-tide line within the range of *E. modestus. Xenostrobus pulex* and *E. modestus* are both known as fouling organisms found also in aquaculture farms where they both settle densely on the rock and Pacific oysters (Menzel, 1991). The ability of the mussel to co-exist with this barnacle depends on favourable spawning (Morton and Miller, 1968). Morton (1999) observed a *X. pulex* community on a south Australian coastline and reported that growth and recruitment was sequentially structured down the shore according to physiological stresses, competitive grazing and predation.

In Tolaga Bay *X. pulex* and the honeycomb barnacle *Chamaesipho columna* were dominant species in constructed habitats, while the barnacle *E. modestus* (a common feature elsewhere) was absent. *E. modestus* preferentially lives in bays and other sheltered areas and these patterns have been observed in other studies (Foster, 1971). Morton and Miller (1968) also studied the Tolaga Bay coastlines and observed that *X. pulex* and *C. columna* dominated midlittoral zones. The little black mussel is likely selectively favoured at this beach due to its ability to withstand sand-scour, while both the little black mussel and honeycomb barnacle do well because of the absence of their predator the oyster borer *Lepsiella scobina.* Tolaga Bay is situated on a stretch of open coast and these habitats, and corresponding wave conditions, are favourable for *C. columna* (Moore, 1944). A major threat to barnacles is predation from starfish, crabs and gastropods, because they are often zoned on shores to meet their adaptations and dietary requirements (Foster, 1987). To overcome the threat posed by predators in the intertidal, *C. columna* will settle in high densities, often termed ‘swamping’, which improves the chances of some individuals reaching breeding age (Foster, 1987). Like *Elminius modestus, C. columna* also has a long breeding period that exists throughout the year (Moore, 1944), and this will support the abundant offspring required to efficiently compete for space.

Kaimoana species in constructed habitats

Artificial structures were found to enhance the abundance of kaimoana species, but they also increased the available space and enhanced populations of non-indigenous species. One of the most abundant species associated with artificial habitats is the native barnacle *Elminius modestus* (discussed above). While Māori do not harvest these prominent species nowadays, there is evidence suggesting that the barnacle *E. modestus*, at least, was collected as a food source by pre-European Māori. Foster (1986) studied ancient Māori middens (a mound of domestic refuse from prehistoric settlement) where he found that *Elminius modestus* was commonly associated with middens that contained cockles and pipi. *Chamaesipho columna* was also abundant at Tolaga Bay and although it is not contemporarily collected for food today, it was also recorded in Māori middens that also contained pipi and mussels (Foster, 1986). The invasive oyster *Crassostrea gigas* was also part of the dominant community found on artificial habitats. Obviously pre-European Māori did not harvest this species of oyster, since it was not present in New Zealand, but nowadays both native and invasive species are both indiscriminately collected for food (Hay and Lindsay, 2003). The native rock oyster (*Saccostrea glomerata*) was found on constructed structures, but was commonly found in natural habitats. This suggests that kaimoana used today are not well represented in constructed habitats, and may not provide useful alternative collection sites.

A look at the rock oyster farming history and reproduction in New Zealand can provide reasons for why the Pacific oyster is more common on constructed sites. The rock oyster was locally and traditionally harvested until New Zealand's government initiated their exploitation in 1877 (National Research Council, 2004). Initially rock oysters were cultivated on rocks in west coast harbours, on rock boulders that were placed in the intertidal regions to provide suitable habitats for their settlement (Menzel, 1991). Predator removal was used as a method to manage natural oyster beds and maximize spat settlement between 1908 and 1960 (National Research Council, 2004). In 1964 oyster farms on leased areas were granted (National Research Council, 2004) and cultivation was on timber sticks that were then positioned in the intertidal (Menzel, 1991). In the 1970s rock oyster production accomplished more than 500 metric tons, but irregular spatfall and competition with the Pacific oyster in 1971 led to the decline in rock oyster cultivation (National Research Council, 2004). There was much debate about which oyster would be more profitable for farming but the Pacific oyster validated its own success by producing heavy spatfalls and growing to a market size in half the amount of time taken by the native rock oyster (Menzel, 1991). From a conservationists point of view it would be important to maintain native biodiversity and farm the rock oyster, but unfortunately from a farmer's perspective it is be more profitable to farm a species that could reach marketable size in the least amount of time.

The establishment of the Pacific oyster is likely the cause for the reduction of rock oysters found on artificial habitats. 'The New Zealand Seashore' by Morton and Miller was published in 1968, their work and collation of marine life is important to this study because it highlights the differences in community assemblages before and after the Pacific oysters establishment. One significant detail they observed was that the rock oyster was dominant on nearly every piling in its range. Pacific oysters are selectively bred in New Zealand and have high fecundity; these two aspects provide the high propagule pressure that would permit them to dominate artificial habitats. My results show that artificial habitats in the Waitemata harbour (Devonport, Bayswater and Birkenhead) have three to four dominant species on each artificial substrate. Pacific oysters dominated communities at Devonport and Bayswater, but the rock oyster was also present on these structures. This suggests that the rock oyster can maintain at least part of the habitat even if it has a lower density than the Pacific oyster. It is often theorized that a highly diverse community will provide biotic resistance from an invasive species. Stachowicz *et al.* (1999) demonstrated experimentally that invasion could be reduced with high biodiversity. They created experimental communities that were composed of zero to four natives, and their results illustrated that as the number of native species increased, the number of surviving invaders decreased.

The non-indigenous Pacific oyster (*Crassostrea gigas*) was the only non-indigenous species recorded during my study, where it resided predominantly on constructed wood and concrete substrates. The non-indigenous barnacle *Balanus trigonus* was observed at Tolaga Bay, but in subtidal regions and therefore was not scientifically quantified. This Pacific oyster is native to Japan, China, Taiwan, Korea and Russia and has established populations in all continents globally except Antarctica (National Research Council, 2004). Smith *et al.* (1986) have proposed that the rapid spread around the northern coasts is through planktonic dispersal and transfer of oyster sticks. Pacific oysters are preferentially farmed in all the east and west coast harbours in the northern part of the North Island, and also in the Coromandel and Ohiwa harbours. Pacific oysters were first discovered in New Zealand in 1971, on the east coast in Mahurangi Harbour. Smith *et al.* (1986) have suggested that three Japanese vessels moored in the Mahurangi harbour were the likely cause for its introduction. Pacific oysters have since invaded most coastal inlets in the North Island, taking only seven years to replace the rock oyster in the farming industry (Menzel, 1991). Reproductive success, available space and competitive displacement can explain their prevalence on these structures and these are explained in further detail.

Reproductive success

It has been suggested that the Pacific oyster is prolific in New Zealand's waters due to its impressive breeding success (Menzel, 1991). It has been calculated that 55 x 10^5 to 1 x 10^8 eggs can be released by the Pacific oyster annually, depending on living conditions (Sato, 1967). During spawning, both reproductive gametes are released into the water column and fertilisation occurs. This effectively allows dispersal of young to find potentially favourable homes, depending on currents. Optimum temperature for development of the Pacific oyster is approximately 23-25 °C, and fertilised eggs require an optimum salinity of 23-28 ‰ (Fujiya, 1970). These requirements are found in New Zealand estuarine environments, and they have the ability to tolerate wide fluctuations in salinity, temperature and turbidity (National Research Council, 2004).

Possible vacant niches

Available space has been experimentally shown to be an important resource that regulates communities (Stachowicz *et al* 1999). Native species co-exist in natural habitats by inhabiting particular niches and partitioning resources such as space amongst their community. The introduction of artificial habitats provides additional space that can be utilised in near-shore communities. Morton and Miller (1968) studied the fauna living on a wharf piling in Devonport Harbour and reported a zonation of species that included *Elminius modestus, Xenostrobus pulex* and *Crassostrea glomerata*. Morton and Miller's (1968) observation of piling fauna is comparable to my study at Devonport, except the non-indigenous Pacific oyster has now become the dominant oyster. Glasby *et al.* (2007) have shown that non-indigenous species in Sydney Harbour, Australia were 1.5-2.5 times greater on artificial habitats than on rocky reefs, even when the local native species pool was greater than that of non-indigenous species. The Pacific oyster may be more efficient at colonising new surfaces compared to the rock oyster. However, Morton and Miller (1968) observed the rock oyster present on pilings before the Pacific oyster's arrival in New Zealand. This suggests that either disturbance (e.g. harvesting) removed rock oysters, thus allowing Pacific oysters to succeed them, or perhaps Pacific oysters are competitively superior and outcompeted the rock oyster in these habitats. It is more likely to be the latter because where the Pacific oyster has invaded habitats, they have been observed to displace native organisms and modify habitats to suit themselves particularly in the south western Atlantic (e.g., Orensanz *et al.,* 2002). The establishment of the Pacific oyster may have initially come about by exploiting vacant niches that were not utilised by the rock oyster. The niches of the two species are known to differ. For example, the rock oyster is tolerant of turbid waters with low salinity, and was observed to flourish on artificial structures in harbours before the introduction of the Pacific oyster (Morton and Miller, 1968). After the arrival of the non-indigenous Pacific oyster there was a common observation that rock oysters occurred in the top portion of pilings due to its ability to tolerate emersion conditions (Dromgoole and Foster, 1983), and Pacific oysters were found lower down (Morton, 2004). In my study I observed the Pacific oyster on artificial habitats but there was some slight overlap at the Raglan site, where I observed and sampled the Pacific oyster on rocky habitats below populations of rock oysters. This is because an important requirement of rock oysters is that it needs to be raised high enough above the reach of settling silt (Morton and Miller, 1968). Morton and Miller (1968) observed that the rock oyster settled more in harbours, and then thinned out in numbers on beaches due to moderate wave energy having the ability to hinder settlement. The ability of the Pacific oyster to live in silt conditions and fill a vacant niche will contribute to its successful invasion. It does require shells (living or dead) for initial settlement on mudflats and soft shores (Diederich, 2005), but then builds up reefs of oysters systematically as the larvae from the next generation settle on adult shells (Lang and Buschbaum, 2009).

Regions dominated by oysters

Unfortunately some sample sites could not be represented in the multivariate analysis because these sites had only single species. In Whangarei inner harbour, concrete and wooden wharf pilings consisted of only Pacific oysters; in contrast the nearby rock reefs were covered in rock oysters. In Manukau harbour similar trends were observed where rock oysters were affiliated with rocky reefs and Pacific oysters with wooden pilings. The concrete pilings in the Coromandel inner harbour consisted of only two species, the first being *Elminius modestus* and the second was the Pacific oyster. Rocky reefs in the Coromandel could not be represented in the results as they also had only single species, but the rock oyster heavily dominated these natural reefs. If both habitats at the Whangarei, Manukau and Coromandel study sites had been included in this investigation they would strongly support the theory that NIS are strongly correlated with artificial structures. These trends have been identified overseas (Klinger *et al.* 2006; Neves *et al.* 2007), whereby invasive species live on artificial substrates and native species tended to associate with naturally occurring substrates. The Whangarei, Manukau and Coromandel study are important for this study because they highlight a trend in that they all represent soft shore habitats where the Pacific oyster was clearly dominant. The artificial structures recorded at the above sites were in soft shores and were predominantly located within the inner harbours.

Natural Rocky Substrates versus Constructed Habitats

The natural rocky reefs I studied had a wide range of species in comparison to their neighbouring artificial habitats. The differences in biodiversity between artificial and natural habitats can be observed in the MDS plot, where artificial habitats clustered close together suggesting similar species, but for rocky reef sites they were spread across the ordination representing a higher variety of assemblages among habitats. I therefore discuss these natural rocky sites separately.

Mount Maunganui

The community composition at Mount Maunganui was comprised of the black dome shaped gastropod (*Nerita atramentosa*), the common barnacle (*Elminius modestus*) and the native rock oyster. The community on natural reefs were more diverse than the community living on nearby wooden pilings. In particular, the black gastropod *Nerita* was an unusual feature at this site, and was only present here on rocky habitats. This herbivorous gastropod belongs to a warm temperate family that specialises in living in the upper shore; its range is along the northern east coast of the North Island, particularly in bays, excluding the Waitemata harbour and it is absent on the west coast (Morton and Miller, 1968). The range of this gastropod explains why I have recorded it in only one of my sites, as all other sites were either on the west coast, in the Waitemata harbour, or too far south.

It is possible that the Pacific oyster could have been present in Mount Maunganui and that people were selectively harvesting the species over the rock oyster. The Pacific oyster grows to twice the size as rock oysters and these would be more appealing to harvesters, but I would likely have seen evidence of their removal because they leave scar marks on the substrate and the cemented bottom valve is often left behind. Rock oysters were highly abundant on these reefs and so it is likely that if any planktonic larvae drifted around these areas, biotic resistance would have repelled Pacific oyster settlement.

Birkenhead

Biotic resistance is the most likely feature that has prohibited settlement at Birkenhead since the Pacific oyster was recorded at two different sites in the same harbour. The rock oyster dominated the rocky shore at Birkenhead and was so abundant that it settled densely on concrete habitats as well. The wooden substrate sampled was a tree trunk that was fully encrusted with the rock oyster, although this was classed as a wooden substrate and therefore included within the artificial habitats. The spotted top-shell, maihi (*Melagraphia aethiops*) and the snakeskin chiton (*Sypharochiton pelliserpentis*) were sampled in relatively equal abundances on rocks in Birkenhead.

Raglan

Both species of oysters were recorded in rocky areas although the rock oyster was the dominant species and was found higher on the shore compared to the Pacific oyster. While the concept of biotic resistance can be extrapolated to the diverse community at Birkenhead, it does not seem to apply at the Raglan site due to the presence of Pacific oysters. There are three possible explanations for its presence in these rocky areas; 1) the rock oyster was the only native species present on the natural substrates, and due to having slightly different niches may not have the capabilities to repel the invasion on its own, 2) the rocks were located in the inner harbour of Raglan, which had silt/mud sediments surrounding the rocks, and 3) rock oysters were more abundant above the low water mark but their distribution did not extend below this boundary possibly due to the presence of the silt sediments.

Whakatane

The rocky shore at Whakatane contains a diverse community assemblage consisting of kina (*Evechinus chloroticus*), cat's eyes (*Turbo smaragdus*), and the green-lipped mussel (*Perna canaliculus*). These three species remain important food sources for Māori, while the green-lipped mussel is also very important to New Zealand's shellfish aquaculture (Westpac mussels distributors). Seeds of *P. canaliculus* contribute >70, 000 tonnes of seed mussels to the aquaculture industry every year, and these seeds are caught from wild mussel populations in the northern part of New Zealand (Alfaro, 2001). The inner harbour of Whakatane is a relatively muddy environment due to the silt load derived from the river sediments. This type of environment could possibly provide suitable habitats for the establishment of the barnacle *E. modestus*, the small black mussel *X. pulex* and the Pacific oyster. The absence of Pacific oyster on rocky reefs, present in constructed habitats here, could be attributed to the diverse community having the ability to resist invasion by occupying available space.

Port Waikato and Gisborne

Rocky reefs at Port Waikato were similar in species to those found in Gisborne, in that they both had the small black mussel (*X. pulex*) and one of the common barnacles, *Elminius plicatus.* Foster (1986) consistently found this barnacle in Māori middens and suggested that its exploitation was to about the same level as that of cat's eyes/ pupu/ ataata. This barnacle is one of the four most common species of New Zealand barnacles in the mid tidal elevations and is generally always below the small black mussel, it is also rare in harbour waters and this may explain why I recorded them at these sites because they are exposed coasts (Morton and Miller, 1968). Biodiversity and species abundance were both greater in Gisborne compared to Port Waikato. I observed that *E. plicatus* was generally higher on the shore compared to *E. modestus*. Morton and Miller (1968) have also observed this common zoning of these two barnacles. The oyster borer *Lepsiella scobina* was relatively abundant at this beach and was closely associated with their barnacle prey.

Tolaga Bay

The barnacle (*C. columna*) dominated intertidal community compositions at Tolaga bay, and distinguished this site from elsewhere, and was closely associated with the small black mussel (*X. pulex*). The community on the rocky shores were identical in species and relative abundance to their nearby concrete habitats. While intertidal species were all native organisms, the subtidal community was completely different and was dominated by the green-lipped mussel and the barnacle *Balanus trigonus.* This barnacle is a non-indigenous species that was introduced to New Zealand in 1960 by hull fouling (Cranfield *et al.* 1998).

Fauna Associated with Green-lipped Mussels

The farmed green-lipped mussel makes up a multi-million dollar aquaculture industry and is the largest single species of seafood exported today. The mussels are farmed using the long-line method, where they are suspended and supported by floats, and is the most efficient and environmentally friendly method available. These mussels are exported both nationally and worldwide either live in the mussel shell or as meat in punnets (Westpac mussels distributors). I investigated the fauna associated with mussels in North Island supermarkets to see whether these mussels were moving additional and unintentional species. *Balanus trigonus* is known to be associated with mussel shells (Foster, 1967) and so this study aimed to identify if this association with mussel shells. The white slipper shell (*Crepidula monoxyla*) is native to New Zealand and was associated with green-lipped mussels in relatively low frequencies.

In some Māori communities it is common to discard mussel, kina and paua shells back into the ocean (often regarded as returning shells) particularly after diving (personal observation). There is a potential for this to be an important vector for introducing and facilitating the spread of this barnacle further around New Zealand's coastline. Although these will typically be cooked and so it is likely that these introductions will have a low chance of establishment. So far this barnacle is known to range from Whangarei and Waitemata harbours, the northeast coast of the North Island through Hauraki Gulf south to Mercury Bay (Cranfield *et al.* 1998).

Conclusion

Marine Reserves

The conclusions from this study suggest that with total protection the intertidal and shallow subtidal populations of kina will increase with time. However, the results also suggest that protection of cat’s eyes may either have no effect, or even negatively impact, this species. The Gisborne reserve supports this theory as more cat’s eyes were counted outside of the protected area. Neither species responded to the partially protection at Mimiwhangata Park. This suggests that for rahui, total protection needs to be put in place for a period of time. A survey for the type of kaimoana preferentially collected in each area would strengthen the conclusions of my results immensely. Kina is a preferred kaimoana that is harvested outside of all three of the reserves, but it is not known whether cat’s eyes are harvested outside two of the reserves. If this species were not selectively collected in the Leigh area, we would not expect to see any differences in the populations living within and outside the reserve. If there were no harvest pressure on cat’s eyes in Mimiwhangata then we would also see the result of no significant difference with reserve status. The examination of two traditionally important species in the intertidal region demonstrates that there are combinations of interrelated processes operating on different spatial scales. Differences in establishment dates of the reserves can offer species a different duration of protection, so in the Leigh reserve the kina is found to be highly abundant within the reserve compared to outside areas. This high density illustrates that kina is positively responding to the protection offered by the reserve, and the high density can be attributed to the 30 years of preservation from harvesting. Differences in the level of protection were also a defining feature that will determine whether or not targeted species will benefit from semi-protected areas. Leigh and Gisborne reserves are both permanently closed off areas from harvesting and kina has responded as intended. Mimiwhangata allows recreational fishing and consequently there are no beneficial gains observed for marine life. In this study there were no significant differences measured for the Mimiwhangata Park and neighboring coastlines. This fact has been observed for other species such as snapper and crayfish (Enderby & Enderby, 2006), which are important predators regulating ecosystem dynamics. The importance of these two species is confirmed in the no-take Tawharanui Marine Park (located near Leigh), where the presence of both species are being replenished and causing a significant shift in trophic dynamics (Shears & Babcock, 2002).

Constructed Structures

Fauna of constructed structures had lower biodiversity compared to nearby natural habitats. The small black mussel *Xenostrobus pulex* and the barnacle *Elminius modestus* are both found in high abundances in artificial habitats, probably due to their reproductive success and the absence of their predator *Lepsiella scobina.* Of the traditionally used species, the native rock oyster (*Saccostrea glomerata*) was found in relatively equal frequencies on artificial habitats and rocky reefs, and the non-indigenous oyster (*Crassostrea gigas*) was predominantly recorded on artificial structures and rare on natural substrates. The construction of artificial structures around New Zealand’s coastline may assist in the spread of non-indigenous species, to the detriment of species traditionally used by Māori.

References

Alfaro, A. C. (2006) Tidal migration influences the zonation of grazing snails (Turbo smaragdus) in a mangrove-seagrass estuary, Northern New Zealand. *Estuaries and coasts* **29**, 731-736.

Alfaro, A. C., Dewas, S. E., Thomas, F (2007) Food and habitat partitioning in grazing snails (Turbo smaragdus), Northern New Zealand. *Estuaries and coasts* **30**, 431-440.

Andrew, N. A. (1988) Ecological aspects of the common sea urchin *Evechinus chloroticus* in northern New Zealand: a review. *New Zealand Journal of marine and freshwater research* **22**, 415-426.

Babcock, R. C., Kelly, S., Shears, N. T., Walker, J. W. and Willis, T.J. (1999) Changes in community structure in temperate marine reserves. *Marine ecology progress series.* **189**, 125-134.

Bando, K.J. (2006) The roles of competition and disturbance in a marine invasion. *Biological Invasions* **8**, 755-763.

Best, E. (1929) Fishing methods and devices of the Maori.

Booth, J. D., and Cox, O. (2003) Marine fisheries enhancement in New Zealand: our perspective. *New Zealand journal of marine and freshwater research.* **37**. 673-90.

Brown, P. J; and Taylor, R. B. (1999) Effects of trampling by humans on animals inhabiting coralline algal turf in the rocky intertidal. *Journal of experimental marine biology and ecology*. **235**, 45-53.

Bucknill, C. E.R., Powell, A.W.B. (1924) Sea shells of New Zealand. Auckland, Whitcombe &Tombs.

Bullock, T. H. (1953) Predator Recognition and Escape Responses of Some Intertidal Gastropods in Presence of Starfish. *Behaviour*. **5**.130-140.

Byers, J. E. (2005) Marine reserves enhance abundance but not competitive impacts of a harvested non-indigenous species. *The ecological society of America*. **86**. 487-500.

Carlton, James. T. 1996. Pattern, process, and prediction in marine invasion ecology. *Biological conservation*. **78**. 97-106.

Chapman, J. W; and Carlton, J. T. (1991) A test for criteria for introduced species: the global invasion by the Isopod *Synidotea laevidorsalis* (Miers, 1881) *Journal of crustacean biology* **11**. 386-400.

Clarke, K.R; and Warwick, R.M. (1994) Change in marine communities: an approach to statistical analysis and interpretation.

Department of Conservation. 2007. Mimiwhangata Coastal Park. Whangarei Conservancy. DOC brochure.

Cranfield, H. J., Gordon, D. P., Willan, R. C., Marshall, B.A. Battershill, C. N., Francis, M. P. Nelson, W. A., Glasby, C. J., and Read, G.B. (1998) Adventive marine species in New Zealand. NIWA Technical report. **34.**

Denny, C. M. and Babcock, R.C. (2004) Do partial marine reserves protect reef fish assemblages?. *Biological conservation*. **116**, 119-129.

Diederich, S. (2005) Differential recruitment of introduced pacific oysters and native mussels at the North Sea coast: coexistence possible? *Journal of sea research.* **53**, 269-281.

Duggins, D, O. (1983) Starfish Predation and the Creation of Mosaic Patterns in a Kelp-Dominated Community. *Ecology*. **64**, 1610-1619.

Enderby, J and Enderby, T. (2006) A guide to New Zealand's marine reserves. New Holland publishers.

Fawcett, M. H. (1984) Local and Latitudinal Variation in Predation on an Herbivorous Marine Snail. *Ecology*. **65**, 1214-1230.

Foster, B.A. 1967. A guide to the littoral balanomorph barnacles of New Zealand. *Tuatara*. **15**.

Foster, B.A. (1987). Barnacle ecology and adaptation. Chapter in: *Barnacle Biology, Crustacean Issues* **5.** Balkema, A.A., Netherlands.

Freeman, D. (2006) Te Angiangi and Te Tapuwae o Rongokako marine reserves: intertidal paua and kina monitoring. Department of Conservation report. 26

Freeman, D. (2008) Thesis: The ecology of spiny lobsters (*Jasus edwardsii*) on fished and unfished reefs. The University of Auckland.

Fujiya, M. (1970) Oyster farming in Japan. *Helgolander wissenschaftliche Meeresuntersuchungen.* **20**. 464-479

Gibson, P. (2005) Maori methods and indicators for marine protection: Ngati Konohi interests and expectations the rohe moana. Department of Conservation report.

Glasby, T. M., Connell, S. D., Holloway, M. G., and Hewitt, C. L. (2007) Nonindigenous biota on artificial structures:could habitat creation facilitate biological invasion? *Marine biology*. **151**. 887-895.

Gollasch, S. (2006) Overview on introduced aquatic species in European navigational and adjacent waters. *Helgol Mar Res*. **60**. 84-89.

Harms, J. (1999) The neozoan *Elminius modestus* Darwin (Crustacea, Cirripedia): possible explanations for its successful invasion in European water. **52**. 337-345.

Hay, Brenda and Lindsay, Val. 2003. A study of the New Zealand farmed oyster industry and the potentia for sustainable Maori economic development. James Henare Maori research centre. Auckland.

Kelly, S., Scott, D., MacDiarmid, A. B., and Babcock R.C. (2000) Spiny lobster, *Jasus edwardsii*, recovery in New Zealand marine reserves. *Biological conservation*. **92**. 359-369.

Klinger, T., Padilla, D. K., and Britton-Simmons, K. (2006) Two invaders achieve higher densities in reserves. *Aquatic conservation: marine and freshwater ecosystems.* **16**.301-311.

Lang, A. C., and Buschbaum, C. (2009) Facilitative effects of the introduced pacific oysters on naative macroalgae are limited by a secondary invader, the seaweed *Sargassum muticum. Journal of sea research* **63**, 119-128.

Levine, J.M. 2000. Species diversity and biological invasions: relating local process to community pattern. *Science*. **288**. 852-854.

McCrone, A. (2001) Visitor impacts on marine protected areas in New Zealand. *Science for conservation*. **173**. Department of Conservation.

Menge, Bruce A. (2000) Top-down and bottom-up community regulation in marine rocky intertidal habitats. *Journal of experimental marine biology and ecology*. **250**. 257-289.

Menzel, W. (1991) Estuarine and marine bivalve mollusk culture. CRC Press. Florida.

Morley, M. S., Anderson, I. A. (2004) A photographic guide to seashells of New Zealand. Auckland, New Zealand, New Holland.

Morton, B. (1999) Competitive grazers and the predatory whelk *Lepsiella flindersi* (Gastropoda: Muricidae) structure a mussel bed (*Xenostrobus pulex*) on a southwest Australian shore. *Journal of molluscan studies*. **65**. 435-452.

Morton, J. (2004) Seashore ecology of New Zealand and the Pacific. Auckland, N.Z: Bateman.

Morton, J., and Miller, M. (1968) The New Zealand sea shore. William Collins sons and Co.Ltd. Great Britain.

National Research Council. (2004) Nonnative oysters in Chesapeake Bay. The national academies press.

Neves, C. S., Rocha, R. M., Pitombo, F. B., Roper, J. J. (2007) Use of artificial substrata by introduced and cryptogenic marine species in Paranagua Bay, southern Brazil. *Biofouling*. **23**. 319-330.

Orensanz, J.M. Schwindt, E. Pastorino, G. Bortolus, A. Casas, G. Darrigran, G. Elias, R. Lopez Gappa, JJ. Obenat, S. Pascual, M. Penchaszadeh, P. Luz Piriz, M. Scarabino, F. Spivak, ED. Vallarino E.A. (2002.) No longer the pristine confines of the world ocean: a survey of exotic marine species in the southwestern Atlantic. *Biological Invasions*. **4**. 115-143.

Povey, A., Keough, M. J. (1991) Effects of trampling on plant and animal populations on rocky shores. *Oikos*. **61**. 355-368.

Romanuk, T. N., and Kolasa, J. (2005) Resource limitation, biodiversity, and competitive effects interact to determine the invasibility of rock pool microcosms. *Biological Invasions*. **7**. 711-722.

Sandrey, R.A. (1986) Maori fishing rights in New Zealand: an economic perspective. Agricultural economics research unit Lincoln College.

Sato, T. (1967) Oyster culture. *Aquaculture*. Tokyo.

Schiel, D. R., and Taylor, D. I. (1999) Effects of trampling on a rocky intertidal algal assemblage in southern New Zealand. *Journal of experimental marine biology and ecology*. **235**. 213-235.

Shears, N. T., and Babcock, R. C. (2002) Marine reserves demonstrate top-down control of community structure on temperate reefs. *Oecologia*. **132**. 131-142.

Shears, N. T., Grace, R. V., Usmar, N. R., Kerr, V. Babcock, R. C. (CS2006) Long-term trends in lobster populations in a partially protected vs. no-take Marine Park. *Biological conservation*. 132. 222-231.

Shinen, J.S., Morgan, S. G., and Chan, A.L. (2009) Invasion resistance on rocky shores:direct and indirect effects of three native predators on an exotic and a native prey species. *Marine ecology progress series*. **378**. 47-54.

Smith, P.J. Ozaki, H. Fujio, Y. (1986) No evidence for reduced genetic variation in the accidentally introduced oyster *Crassostrea gigas* in New Zealand. *New Zealand journal of marine and freshwater research*. **20**. 569-574.

Stachowicz, J. J., Whitlatch, R. B., and Osman, R. W. (1999) Species diversity and invasion resistance in a marine ecosystem. *Science*. **286.**

Westpac Mussels Distributors Limited. (2007) 19 Corban Avenue, Henderson, Auckland, New Zealand. Viewed on 18 February 2010. http://westpacmussels.com/

Walsby, J., Morton, J. (1982) Marine Molluscs, Part 1. University of Auckland, Leigh Marine Laboratory.

Williams. (2006) Resource management and Maori attitudes to water in Southern New Zealand. *New Zealand geographer*. **62**. 73-80.