Marine Reserves
as Conservation and Management Tools:
Implications for Coastal Resource Use

by

Tyler D Eddy

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TE WHARE WĀNANGA O TE ŪPOKO O TE IKA A MĀUI
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Demoiselle (*Chromis dispilus*) at the Meyer Islands, Kermadec Islands Marine Reserve, New Zealand

"Scientific innovation sometimes sounds like poetry,
and I would claim that it is, at least in the earliest stages.
The ideal scientist can be said to think like a poet,
work like a clerk, and write like a journalist.
The ideal poet thinks, works and writes like a poet.
The two vocations draw from the same
subconscious wellsprings and depend upon
similar primal stories and images."

"Biophilia" - Edward O Wilson

This thesis was conducted under the supervision of:

Prof Jonathan PA Gardner (*Primary Supervisor*)

Victoria University of Wellington

Wellington, New Zealand

and

Dr James J Bell (*Secondary Supervisor*)

Victoria University of Wellington

Wellington, New Zealand

Abstract

Exploited marine ecosystems are a common feature of the modern world and area closures (marine reserves; MRs) have been suggested from both conservation and fishery management perspectives as a technique to rebuild over-fished populations. MRs provide an interesting experimental treatment where humans are excluded from resource harvesting. In the absence of exploitation, marine species have been observed to return to levels of abundance similar to historic accounts of virgin biomass (biomass under an exploitation level of 0). This thesis investigates the impact and potential of MRs in both New Zealand and Chile for achieving conservation and fishery management goals through the use of underwater observation, historic information, fishers' ecological knowledge (FEK), bioeconomic fishery modeling and ecosystem modeling.

New Zealand has been inhabited by humans for an estimated 800 years with archaeological evidence of a high dependence on marine resources for subsistence. One region of New Zealand territorial waters that have been largely spared from human exploitation are the Kermadec Islands, which have recently been identified as one of twenty remaining pristine marine ecosystems left in the world. The Kermadec Islands MR is the largest in New Zealand protecting species endemic to the archipelago and species not found elsewhere within the country. I surveyed reef fishes for size and abundance via underwater observation at three sites around Raoul Island and the Meyer Islets and calculated biomasses of trophic groups. Planktivores dominated trophic group abundance at all three sites. This

research represents the first observations of all trophic groups of reef fishes since implementation of the Kermadec Islands MR in 1990 and documents a pristine baseline in the absence of human exploitation.

The Kapiti MR was established in 1992, protecting 2167 ha of coastal marine environment and is located in the Cook Strait region of central New Zealand. I monitored reef fishes at protected and unprotected sites using underwater observation and compared results with previous studies to determine time scales for direct (response of species targeted by humans) and indirect (response of species not targeted by humans) effects. Analysis of monitoring data has highlighted the importance of baseline studies, as one of the protected sites showed a high abundance of reef fishes prior to MR implementation. Results indicate that timelines are variable and species-specific for the response of size, abundance and biomass of targeted and non-targeted reef fishes to MR protection.

The Taputeranga MR, located on the south coast of the capital city of Wellington, New Zealand was implemented in 2008 and protects 854 ha. The location of the Taputerana MR in the Cook Strait means that the marine environment it protects is subject to high wind, wave and current energies. Commercial, recreational and traditional fisheries are important in this region and have been documented over the last 70 years and it has been estimated that lobster biomass is now approximately one quarter of its former state. I have used an ecosystem-based modeling approach to analyse food web linkages for the Taputeranga MR at present time for comparison to historic and future ecosystem states. Construction of an Ecopath with Ecosim (EwE) model involved collection of biomass data for all species of marine organisms found in the Taputeranga MR from a variety of sources. This involved my own underwater observations of reef fishes, which is a part of the ongoing Taputeranga MR monitoring program. The role of lobster within the ecosystem has changed from historic times to present as it previously had a much higher keystone role (organising of other species) within the ecosystem than it currently experiences and its diet is predicted to have been more herbivorous in historic times. My results indicate that the Taputeranga MR that is now in place is capable of restoring the future ecosystem to a state more similar to that observed during historic times.

Historic accounts of lobster (*Jasus frontalis*) on the Chilean Juan Fernández Archipelago indicate a high abundance at all depths (intertidal to approximately 165 m), but presently lobster are found almost exclusively in deeper regions of their natural distribution. FEK tells a story of serial depletion in lobster abundance at fishing grounds located closest to the fishing port with an associated decline in catch per unit effort (CPUE) throughout recent history. I have re-constructed baselines of lobster biomass throughout human history on the archipelago using historic data, the fishery catch record and FEK to permit examination of the potential effects of MRs, effort reduction and co-management (stewardship of catch) to restore stocks. I employed a bioeconomic model using FEK, fishery catch and effort data, underwater survey information, predicted population growth and response to MR protection (no-take) to explore different management strategies and their trade-offs to restore stocks and improve catches. My findings indicate that increased stewardship of catch coupled with 30% area closure (MR) provides the best option to reconstruct historic baselines. Based on model predictions, continued exploitation under the current management scheme is highly influenced by annual fluctuations and unsustainable. I propose a community-based co-management program to implement a MR in order to rebuild the lobster population while also providing conservation protection for marine species endemic to the Archipelago.

Overall, this thesis has investigated the effects of human coastal resource use in New Zealand and Chile from social, economic and ecological perspectives through the use of different techniques by synthesising both quantitative and qualitative information sources. MRs are a valuable tool from conservation, management and scientific perspectives as they can rebuild overexploited stocks and return the ecosystem to a more historic state. MRs also provide an understanding of the interaction between coastal resource use and ecosystem-wide changes, which is a crucial element for ecosystem-based management. This thesis has illustrated the importance of comparing present stock biomasses to historic baselines to understand the impacts of exploitation of coastal resources on marine ecosystems.

Dedicated to my mother and father; Claire and Danny

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Eddy TD (2009) New Zealand's Largest and Most Isolated Marine Reserve: the subtropical Kermadec Islands. JMBA Global Maine Environment 9: 8-9.

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Articles in Review/Preparation

Pérez-Matus A, Ramírez FM, **Eddy TD**, Cole R (In review) Community structure at the Juan Fernández Islands: The need to conserve fish faunas.

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> *Author contribution:* T.D.E. conducted the research, wrote the manuscript and presented the oral paper. J.P.A.G edited the manuscript.

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Chapter 1

General Introduction

It has been shown that overfishing is the most important anthropogenic contributor to the ecological extinction of coastal ecosystems (Jackson et al. 2001). Following the commercial extinction of one of the world's most productive fisheries in 1992; the Atlantic Canadian cod (*Gadus morhua*) on the Grand Banks, it was discovered that fishery harvest had been exceeding the sustainable yield since 1962, primarily attributed to improper management calculations of fishing mortality and fish abundance (Hutchings and Myers 1994, Myers et al. 1997). With current fishing effort available to mobilise globally in the face of favourable market prices, it is being observed that human-dominated marine ecosystems are losing biodiversity and limiting the capacity of ecosystems to recover from perturbing events (Berkes et al. 2006, Worm et al. 2006).

As targeted and by-catch species become rare, other species are harvested in their place which has led to a phenomena of lower trophic level fishes being targeted, leading to diminishing catches (Pauly et al. 1998). Management practices that incorporate ecosystem-based strategies and directives are being advocated in order to quantify important ocean functions provided by biodiversity with higher trophic levels represented. (Pitcher 2001; Browman et al. 2004). MRs are one strategy that offer protection from exploitation to a wide range of species.

Currently there are 34 MRs in New Zealand and under the New Zealand Marine Reserves Act (1971), in order for an area to become a MR it must exhibit at least one characteristic of a) uniqueness, b) representative of a region or ecotype, c) distinctive or beautiful, d) be in the national interest. It has been shown that MRs increase abundance and average individual size of many fish and shellfish species, (Cole et al. 1990, Cole and Keuskamp 1998, Cole et al. 2000, Kelly et al.

2000, Davidson et al. 2002, Willis et al. 2003, Denny and Babcock 2004, Okey et al. 2004, Shears et al. 2006, Pande et al. 2008) which has implications for predators and prey of such species. The network of MRs being developed in New Zealand encompasses many social aspects as the public reacts to limited fishing opportunity and whether exploited stocks outside of the reserve will recover (Towns and Ballantine 1993).

Studies of MRs in Chile indicate that the exclusion of anthropogenic activities such as fishing and resource harvest have dramatic effects on the structure of marine communities (Castilla 1999). In central and southern Chile, no-take zones resulted in an increase in the abundance of the keyhole limpet, (*Fissurella sp*.) which led to a decrease in mid-intertidal algae resulting in major food-web modifications (Castilla 1999). Witman and Smith have found that rapid increases of diversity and high biomass turnover at an upwelling site at the Galapagos Marine Reserve may be due in part to the protection of invertebrate predators that consume competitively dominant species, opening up space for inferior competitors (2003).

Chassot et al. have constructed a model of trophic interactions to determine ecosystem response to fishing and found that biological production functions are highly dependent on predation parameters and vary differently according to trophic level (2005). Fishing was shown to modify biomass distribution of the system and strongly affects higher trophic levels more sensitive to exploitation. It was concluded in this model that trophic relationships were important factors in determining how the ecosystem responded to exploitation. A trophic-level based model constructed by Gascuel determined that high fishing pressure at low trophic level of first catch may lead to severe depletions in fish stock, without overfishing being observed (2005). Gascuel also observed that biomass transfer efficiencies, trophic flow rate, and trophic structure regulation (bottom-up vs. top-down) are key factors in determining how an ecosystem will respond to fishing pressure (2005). Okey et al. have developed a trophic model of a Galapagos rocky reef system using Ecopath with EcoSim (2004). One modelling scenario varied the area of protection to determine that if 23% of total reef area was excluded from fishing, the functional extinction of the sea cucumber (*Stichopus fuscus*) known as pepino, predicted to occur without protection, was prevented (Okey et al. 2004).

Bioeconomic fishery models are a technique that can be used to determine relationships between fishing effort and stock abundance or biomass. Models created using the STELLA program incorporating biological processes of an ecosystem such as growth rate, energy transfer rates, mortality rates (due to predation, fishing, and natural causes), immigration/emigration with social and economic factors such as fuel cost, gear efficiency and selectivity, market price, and amount of fishing effort to determine how management strategies such as MRs and fisheries regulations impact the ecosystem (Ruth and Lindholm 2002).

Chile, a distant south-Pacific neighbour, has a markedly different coastal resource management system in place through the use of “caletas”, groups of individuals comprised of fishers and community members responsible for the stewardship of a specific region. This system came into effect after commercialisation and international export of shellfish species led to population crashes in the late 1980’s (Castilla and Fernandez 1998). Two features of the system that allow it to deal with problems associated with conventional management are: an increased understanding in ecological processes and a piece of government legislation conceived in 1991 know as the Chilean Fishing and Aquaculture Law (Castilla and Fernandez 1998). This law legalises the use of community-owned fishing grounds known as management areas (MAs) and transfers stewardship and management of fishing grounds to caletas. Under this law, fishers must choose one area to fish in and are not permitted to fish in other areas, alleviating a previous problem where an area was overexploited and abandoned (Meltzoff et al. 2002). “Caletas” consult with ecologists, tourist developers, government managers, and the Navy in order to make decisions about coastal resource use (Meltzoff et al. 2002). The management of species such as the gastropod “loco” (*Concholepas concholepas*) and scallops (*Argopecten purpuratus*) are two species that are being sustainably managed using this system following a period of overexploitation (Castilla and Fernandez 1998). This type of system will be adopted by fishers if they see it as an option to provide an economically profitable harvest with increasing long-term production, however costs associated with stewardship must be clearly outweighed (Meltzoff et al. 2002).

The Chilean inshore management approach is bottom-up in nature, with fishers and ecologists having more control of coastal resources. In Chile, it has been found that compliance is greater in a community-managed system where local stakeholders have a vested economic interest in the welfare of the resource (Castilla and Fernandez 1998). This system is augmented by MRs, which give an indication of how ecosystem dynamics change in the absence of human exploitation, however at the moment there are only three MRs in Chile.

The overall aim of this thesis is to investigate how MRs perform as conservation and managements tools. This evaluation employs case studies across the subtropical and temperate waters of the South Pacific, in both New Zealand and Chile, spanning gradients of biogeographic community and exploitation history. Chapter two documents reef fishes at the pristine marine ecosystem of the isolated and subtropical Kermadec Islands, which is protected by New Zealand’s largest MR. The third chapter examines the effects of 18 years of MR implementation on size, abundance and biomass of reef fishes at the Kapiti MR across sites of varying habitat quality. The fourth chapter develops an ecosystem model quantifying trophic linkages for the Taputeranga MR for historic and present time periods and employs a scenario to determine if the MR can return the future ecosystem to historic state. Chapter five examines an artisanal lobster fishery on the Juan

Fernández Archipelago and documents shifting baselines in lobster abundance over 400 years of exploitation and assess the performance of MRs and other fishery management tools to reconstruct historic lobster stocks. By investigating the effects of MR implementation on targeted and non-targeted species, ecosystems and fisheries using a variety of methods and techniques, across the disciplines of ecology, economics, history and social sciences this thesis provides insight about the potential of MRs for conservation and management perspectives.

The specific aims of this thesis are fourfold: to quantify size, abundance, biomass and trophic structure of reef fishes in the pristine, subtropical Kermadec Islands MR in comparison to protected and exploited ecosystems in temperate New Zealand waters at the Kapiti and Taputeranga MRs. Secondly, to construct models that represent the lobster fishery at the Juan Fernández Archipelago, Chile and the marine ecosystem structure of the Taputeranga MR. Thirdly, to compare historic levels of lobster abundance Juan Fernández Archipelago and ecosystem state at the Taputeranga MR to present day in order to determine the degree of exploitation as well as to predict population and ecosystem trajectories of response under MR management schemes. Lastly, to synthesise the observations and predictions of MRs across a range of biogeographic and exploitation history gradients in order to evaluate their potential as conservation and management tools.

Chapter 2

Abundance and Trophic Structure of Reef Fishes at the Kermadec Islands Marine Reserve, New Zealand

2.1 Abstract

The Kermadec Islands have been identified as one of the few remaining pristine marine ecosystems left in the world. The Kermadec Islands Marine Reserve (MR) is the largest in New Zealand protecting species endemic to the archipelago and species not found elsewhere within the country. I surveyed reef fishes for size and abundance at three sites around Raoul Island and the Meyer Islets and calculated biomasses of trophic groups. Planktivores dominated trophic group abundance at all three sites. This research represents the first observations of all trophic groups of reef fishes since implementation of the Kermadec Islands MR in 1990.

2.2 Introduction

The isolated Kermadec Islands (29 - 31°S, 178°W) located 750 km northeast of Cape Reinga, represent the only true subtropical marine habitat in New Zealand. The volcanic Kermadec Island archipelago is composed of four main island groups; Raoul Island and surrounding Herald Islets in the north, Macauley and Haszard Islands, Curtis and Cheeseman Islands and the southernmost island group of L'Esperance and Havre Rocks (Figure 2.1).

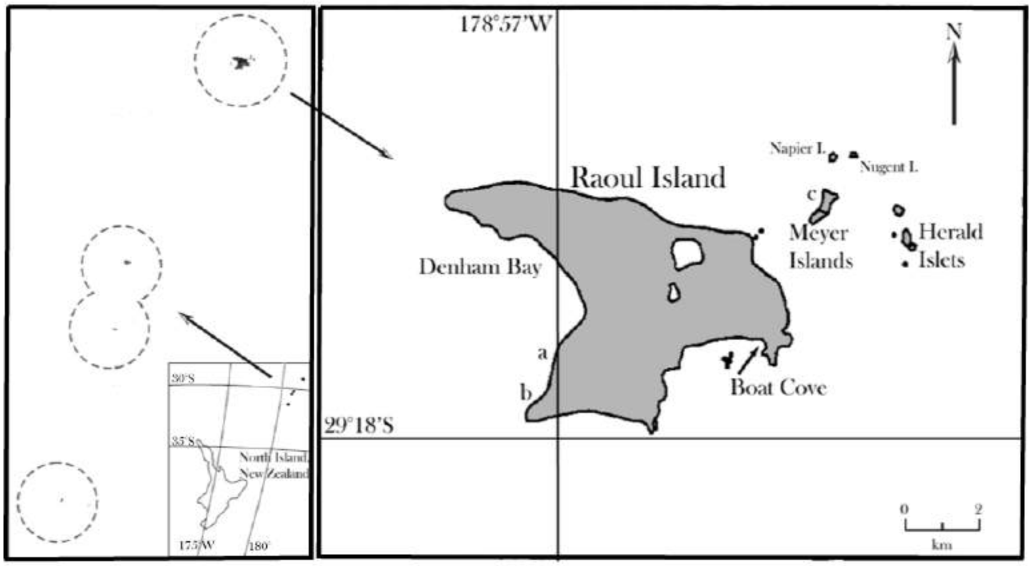


Figure 2. 1. Kermadec Islands Marine Reserve boundaries (left panel, dotted lines), location of Kermadec Islands relative to New Zealand (insert) and map of Raoul Island and surroundings with survey locations (a = Denham 1; b = Denham 2; c = Meyer). Island groups in the left panel from north to south: Raoul Island and surrounding Herald Islets; Macauley and Haszard Islands; Curtis and Cheeseman Islands; L'Esperance and Havre Rocks. Kermadec Islands Marine Reserve map (left panel) modified with permission from the New Zealand Department of Conservation.

Located between between New Zealand (34°S) and Tonga (21°S), the Kermadec Islands harbour a mix of temperate and tropical species (Schiel et al. 1986, Francis et al. 1987, Francis 1991, Cole et al. 1992, Francis 1993, Brook 1998, 1999, Cole 2001, Gardner et al. 2006, Wicks et al. 2010). Sea surface temperature varies from 18 to 24°C seasonally (Francis et al. 1987). While corals are present, both coral reefs and macroalgal stands are absent at the Kermadec Islands (Schiel et al. 1986, Brook 1999).

Along with early species checklists, subtidal research at the Kermadec Islands has been limited due to geographic isolation and rapidly changeable weather conditions. Schiel et al. (1986) conducted fish transects at Boat Cove, located on the south-eastern side of Raoul Island and reported an absence of some tropical herbivores in the scarid, acanthurid and siganid families, whereas families found in temperate waters were well-represented. The vertical distribution of herbivorous fishes at the Kermadec Islands is more similar to temperate waters than tropical as herbivores are more abundant in shallow transects, corresponding to the highest abundance of turfing algae as large stands of macroalgae are absent (Cole et al. 1992). In 1992, the first survey to quantify abundance of fishes since implementation of the MR indicated that herbivorous fish were mostly represented by one species at Denham Bay, whereas more than one species represented herbivores at Meyer Island (Cole 2001).

Benthic primary producers are comprised of scleractinian corals which are found at shallow depths as well as foliose, filamentous and encrusting red algae. Notably absent are fucalean and laminarian algae. Grazing invertebrates are represented by echinoderms such as the crown of thorns (*Acanthaster planci*) and gastropods such as the endemic giant limpet (*Patella kermadecensis*). Herbivorous fishes are represented by grey and caramel drummer (*Kyphosus bigibbus* & *Girella fimbriata*) and the Pacific gregory (*Stegastes fasciolatus*). Demoiselle (*Chromis* *dispilus*), Blue maomao (*Scorpis violaceus*), Orange wrasse (*Pseudolabrus luculentus*) and Mimic blenny (*Plagiotremus tapeinosoma*) make up the balance for the most observed species. Larger fish species are represented by the spotted-black grouper (*Epinephelus daemelii*), kingfish (*Seriola lalandi*), northern kahawai (*Arripis xylabion*) and Galapagos shark (*Carcharhinus galapagensis*).

Francis et al. (1987) reported that the fish fauna of the Kermadecs is more similar to that of Lord Howe Island (31.5°S and 159°E) than of mainland New Zealand (only 20 % of species observed were common or abundant in northern New Zealand waters) as the diversity of fishes decreases eastward from Lord Howe Island (433 species) to Norfolk Island (29°S and 167.5°E; 254 species) to the Kermadec Islands (145 species) and increases northward within the archipelago (Francis 1991, Francis 1993, Francis and Randall 1993).

During the time of initial studies by Francis et al. (1987) in 1984 and 1985, long line fishing pressure in New Zealand was beginning to expand geographically following declines of hapuku (*Polyprion oxygeneios*) and bass (*Polyprion americanus*) stocks with fishing trips planned to the Kermadecs. Subsequently Francis submitted an application (1985) for a marine reserve (MR) to protect New Zealand's only subtropical marine ecosystem and the Kermadec Islands MR was designated in 1990. It is New Zealand's largest MR at 748 000 ha and extends 22 km seaward from all four island groups (Figure 2.1). The Kermadec Islands MR protects species endemic to the Kermadec Islands as well as highly targeted commercial species in a region where recruitment occurs locally for some species (Francis et al. 1987). The Kermadec Islands have recently been identified by the Census of Marine Life Project as one of 18 pristine sites that exist globally. There is also a recent initiative by conservation groups; Pew Environment Group - Global Ocean Legacy, WWF – New Zealand and the Royal Forest and Bird Protection Society of New Zealand to increase area of protection by creating a Kermadec Ocean Sanctuary.

The aim of this study is to provide a snapshot of reef fish abundances and trophic structure at three sites around Raoul Island and the Meyer Islets. While Cole studied herbivorous fishes post-implementation of the Kermadec Islands MR (2001), this study represents the first observations of all reef fish species since implementation of the Kermadec Islands MR in 1990. Knowledge the abundance, biomass and trophic structure of reef fishes at the Kermadec Islands MR is a valuable link to understand the geographical transition of reef fish assemblages from New Zealand's temperate to subtropical waters.

2.3 Methods

Fish size and abundance surveys were conducted using SCUBA over five days from March 25th until March 29th, 2008, at three sites inside the Kermadec Islands MR. Two sites located on the south-western side of Raoul Island (south of Denham Bay and referred to as Denham 1 – D1; Denham 2 – D2) were approximately 2 km apart and a third site was sampled on the western side of the Meyer Islands (referred to as M; Figure 2.1). Sample site availability was largely determined by the prevailing north-easterly winds, which limited access to other areas. Denham Bay is characterised by large boulders, rocks, cobble and sand with a gentle slope from the intertidal to subtidal zone (Brook 1998). The Meyer Islands are characterised by vertical walls with habitat features such as caves, overhangs and crevices (Brook 1998). All surveys were conducted on rocky reef substratum between 5 and 25m in depth. Sea surface temperature was recorded as 24°C for all surveys in this study, which is the annual high.

Underwater surveys were undertaken using timed counts in a 5 x 5 m^2 area from the sea floor to the surface, which was observed for three minutes (hereafter referred to as a 'station') in order to record both benthic and pelagic fishes. This method was chosen to sample a range of depths with limited time. Individual fishes that were obviously attracted to the diver from outside the survey area were not recorded. Stations were separated by 20 m intervals and were sampled perpendicular to the shore to include a range of depths representative of nearshore subtidal rocky reef habitat at each site (n = 10 stations at D1; n = 32 stations at D2; n = 13 stations at M). Denham 1 was characterised by a shallow, gently sloping shelf; Denham 2 by a steeper sloping shelf; Meyer was characterised by steeper vertical walls rising from a deeper shelf. At each station, all fishes inside the survey area were identified, counted and visually estimated for size class to the nearest 5cm interval. Fish species were assigned to one of five trophic groups; herbivores; omnivores; planktivores; benthic invertebrate feeders; fish and invertebrate feeders (Francis 2001; Froese and Pauly 2005). Size-frequency data were converted into biomass using non-linear length to weight relationships for north-eastern New Zealand reef fishes as described by Taylor and Willis (1998) and FishBase (Froese and Pauly 2005). A species accumulation curve was generated using data pooled across all three sites with PRIMER software using the bootstrap technique with 9999 permutations (Clarke & Gorley 2006).

2.4 Results

Thirty-eight species of fish were recorded in the timed counts and a further three species were sighted during the expedition; *Pterois volitans* (lionfish), *Canthigaster callisterna* (clown toado) and *Evistias acutirostris* (striped boarfish) (Table 2.1). These forty-one observed fish species represented 36 genera, 23 families, seven orders and two classes. Of the 4220 fish counted during underwater surveys, the ten most abundant species accounted for 93 per cent of all fish (Figure 2.2). The highest mean number of species per station was observed at D2 (7.2 ± 0.2

species; mean ± SE), whereas M showed the highest fish abundance and biomass per station (114.9 ± 28.5 individuals; 39. 7 ± 16 kg respectively). The mean size of *Chromis dispilus* was larger at M (15 ± 2 cm; mean ± SE) than D1 and D2 (D1 = 7 ± 1 cm; D2 = 8 ± 2 cm). *Kyphosus bigibbus* showed greatest average size at M (31 ± 4 cm ; D2 = 24 ± 5 cm; mean ± SE) and was not observed at D1. *Pseudolabrus luculentus* was larger at D2 (12 ± 1 cm; mean ± SE) than at D1 (8.8 ± 2 cm) and M (7.1 ± 1 cm).

Planktivores were the most abundant trophic group at all sites, accounting for 71% of total observations, followed by invertebrate feeders (10%), omnivores (8%), fish and invertebrate feeders (7%) and herbivores (4%) (Figure 2.3). Planktivores accounted for 47% of total biomass, followed by omnivores (35%), invertebrate feeders (8%), fish and invertebrate feeders (7%) and herbivores (3%; Figure 2.3). The high biomass of omnivores observed at M was due to a high abundance of *Girella cyanea* (Figure 2.3). Increased sampling effort of reef fishes would likely record new species, as a species accumulation curve did not reach an asymptote (Figure 2.4).

Table 2. 1. Mean individual fish species abundance with trophic group (H = herbivore, P = planktivore, O = omnivore, I = invertebrate feeder, F = fish and invertebrate feeder), total number of species, total abundance and total biomass (kg) per station with standard error (SE) at each site (D1 = Denham 1, n = 10; D2 = Denham 2, n = 32; M = Meyer, n = 13). Table is found on next page.

			Denham 1		Denham 2		Meyer		Trophic Group
Family	Species	Common name	$\bar{x}$	SE	$\bar{x}$	SE	$\bar{x}$	SE	
Carcharhinidae	*Carcharhinus galapagensis*	Galapagos reef shark	0.00	0.00	0.03	0.03	0.00	0.00	F
Aulostomidae	*Aulostomus chinensis*	trumpetfish	0.00	0.00	0.00	0.00	0.15	0.15	F
Trachichthyidae	*Optivus elongatus*	slender roughy	0.00	0.00	1.00	0.69	0.00	0.00	P
Serranidae	*Acanthistius cinctus*	yellow-banded perch	0.00	0.00	0.26	0.09	0.00	0.00	F
	Aulacocephalus temmincki	gold-ribbon grouper	0.00	0.00	0.58	0.47	1.85	1.53	I
	Epinephelus daemelii	spotted black grouper	0.00	0.00	0.10	0.06	0.15	0.15	F
	Trachypoma macracanthus	toadstool grouper	0.00	0.00	0.06	0.04	0.00	0.00	F
Carangidae	*Seriola lalandi*	kingfish	0.00	0.00	0.10	0.05	0.15	0.10	F
Arripidae	*Arripis xylabion*	northern kahawai	0.00	0.00	0.03	0.03	0.00	0.00	I
Mullidae	*Parupeneus spilurus*	black-spot goatfish	0.00	0.00	0.19	0.10	0.23	0.17	I
Pempheridae	*Pempheris analis*	bronze bullseye	0.00	0.00	0.77	0.63	0.00	0.00	P
Kyphosidae	*Kyphosus bigibbus*	grey drummer	0.00	0.00	2.06	1.87	5.92	3.07	O
Girellidae	*Girella cyanea*	bluefish	0.00	0.00	0.00	0.00	9.23	5.25	O
	Girella fimbriata	caramel drummer	0.00	0.00	0.00	0.00	0.08	0.08	H
Scorpidae	*Labracoglossa nitida*	blue knifefish	0.00	0.00	2.90	1.97	0.00	0.00	P
	Scorpis violaceus	blue maomao	0.00	0.00	26.45	6.38	0.00	0.00	P
Microcanthidae	*Atypichthys latus*	mado	0.00	0.00	2.71	1.10	0.00	0.00	O
Chaetodontidae	*Amphichaetodon howensis*	Lord Howe coralfish	0.10	0.10	0.00	0.00	0.23	0.17	I
Cirrhitidae	*Notocirrhitus splendens*	splendid hawkfish	0.00	0.00	0.03	0.03	0.00	0.00	I
Aplodactylidae	*Aplodactylus etheridgii*	notch-head marblefish	0.10	0.10	0.03	0.03	0.00	0.00	H
Latridae	*Cheilodactylus ephippium*	painted moki	0.00	0.00	0.03	0.03	0.00	0.00	I
	Cheilodactylus francisi	masked moki	0.00	0.00	0.03	0.03	0.00	0.00	I
Pomacentridae	*Chromis dispilus*	demoiselle	26.00	10.56	11.13	2.67	51.69	24.70	P
	Chrysiptera rapanui	Kermadec demoiselle	28.60	5.02	8.58	1.67	13.46	2.85	P
	Parma alboscapularis	black angelfish	0.40	0.16	0.69	0.15	0.00	0.00	H
	Parma kermadecensis	Kermadec scalyfin	0.00	0.00	0.13	0.06	1.54	0.33	H
	Stegastes fasciolatus	Pacific gregory	2.30	0.30	3.13	0.77	0.00	0.00	H
Labridae	*Anampses caeruleopunctatus*	blue-spotted wrasse	0.40	0.16	0.39	0.13	0.38	0.24	I
	Anampses elegans	elegant wrasse	0.30	0.21	0.16	0.07	0.62	0.27	I
	Coris sandageri	Sandager's wrasse	1.10	0.41	0.32	0.09	1.46	0.50	I
	Notolabrus inscriptus	green wrasse	0.10	0.10	0.29	0.16	0.08	0.08	I
	Pseudolabrus luculentus	orange wrasse	10.30	3.48	2.03	0.45	7.31	2.14	I
	Thalassoma lutescens	sunset wrasse	0.10	0.10	0.23	0.09	0.08	0.08	I
	Thalassoma trilobatum	ladder wrasse	0.00	0.00	0.19	0.08	0.00	0.00	I
Blennidae	*Cirripectes alboapicalis*	white-dot blenny	0.10	0.10	0.00	0.00	0.00	0.00	H
	Plagiotremus tapeinosoma	mimic blenny	4.40	2.27	5.84	1.41	2.38	1.21	F
Monacanthidae	*Thamnaconus analis*	morse-code leatherjacket	0.30	0.15	0.03	0.03	0.15	0.10	I
Zanclidae	*Zanclus cornutus*	Moorish idol	0.00	0.00	0.00	0.00	0.15	0.15	O
Number of species			6.5	0.4	7.2	0.4	6.7	0.3	
Total abundance			74.6	15.3	68.9	6.9	97.4	26.3	
Total biomass (kg)			3.4	0.8	17.4	3.4	39.7	16.0	

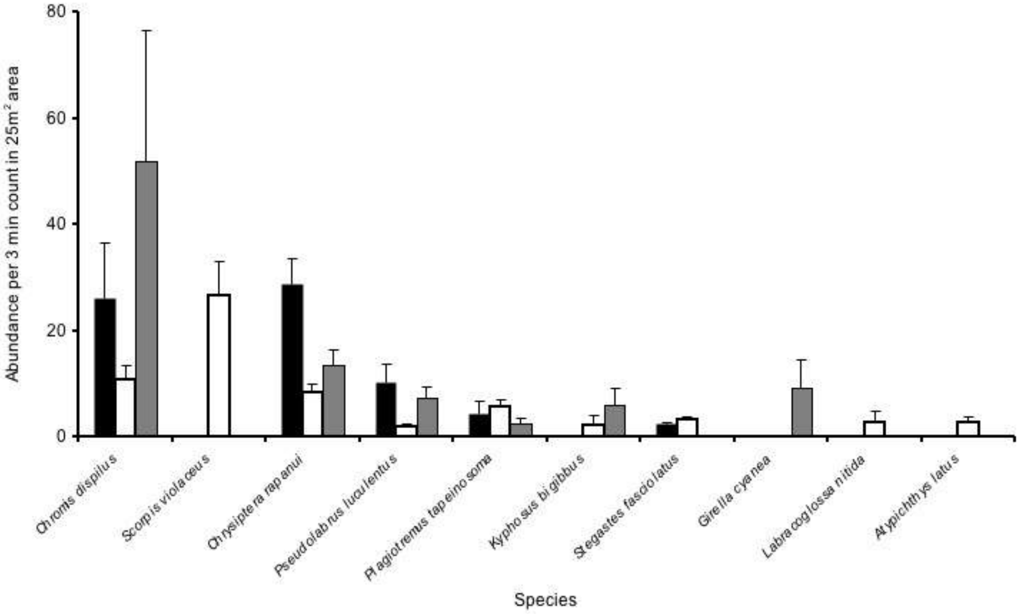


Figure 2. 2. Abundance of ten most frequently observed reef fish species (mean + SE) at Kermadec Islands Marine Reserve survey sites (D1 = Denham 1, black bars, n = 10; D2 = Denham 2, white bars, n = 32; M = Meyer, grey bars, n = 13). Refer to Figure 1 for location of survey sites.

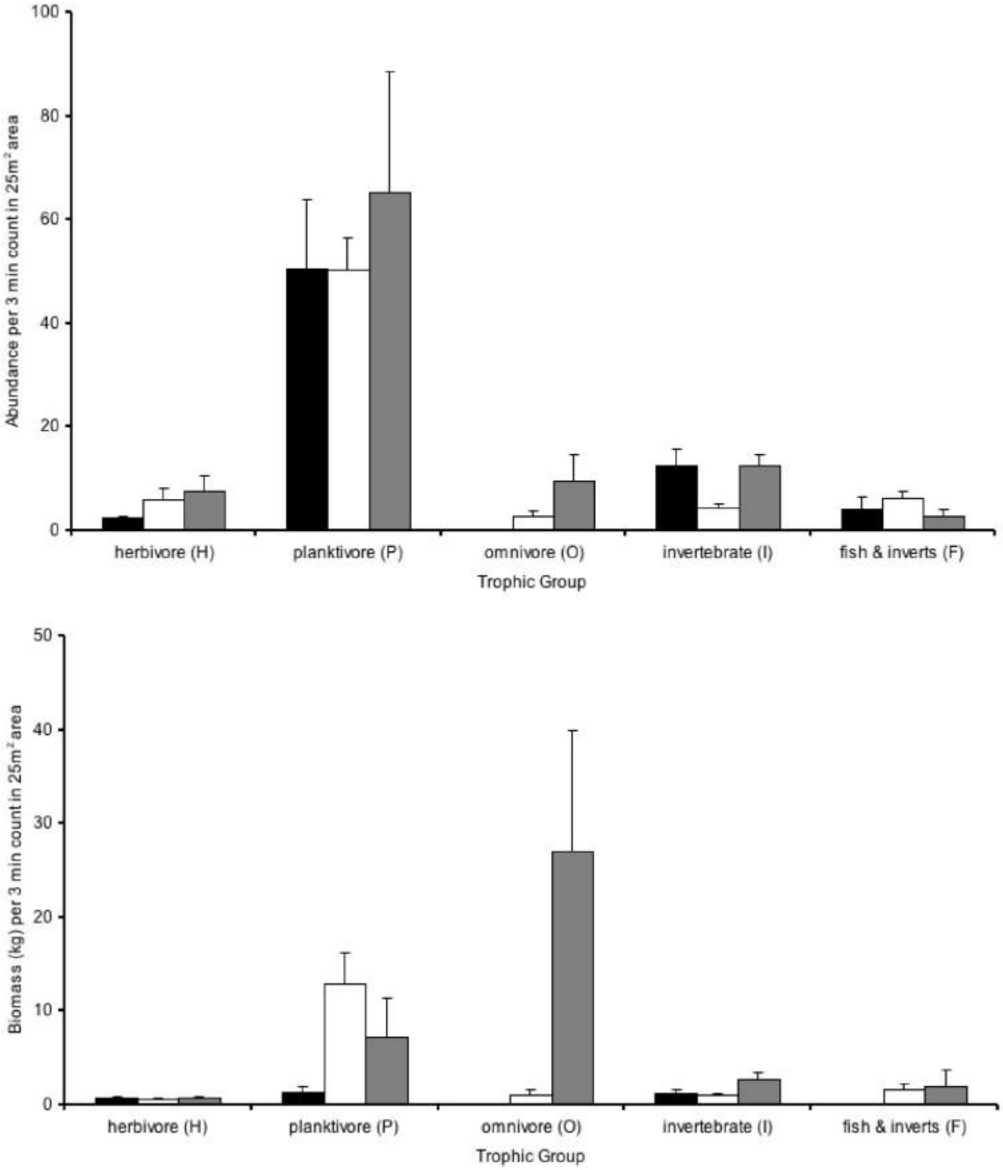


Figure 2. 3. Abundance and biomass (kg) of reef fishes representing each trophic group (mean + SE) at Kermadec Islands Marine Reserve survey sites (D1 = Denham 1, black bars, n = 10; D2 = Denham 2, white bars, n = 32; M = Meyer, grey bars, n = 13). Refer to Figure 1 for location of survey sites.

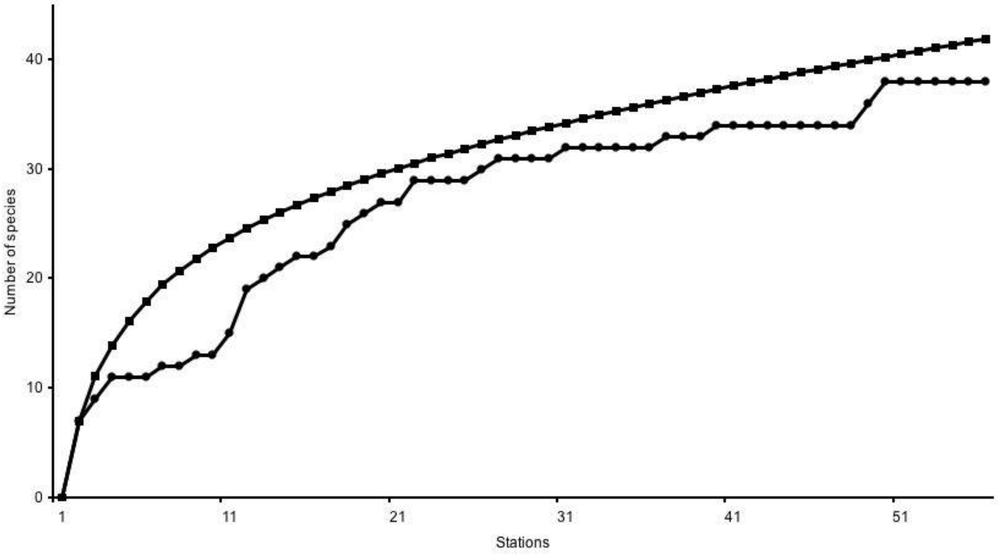


Figure 2. 4. Species accumulation curve showing cumulative number of reef fish species recorded at Kermadec Islands Marine Reserve (data pooled across all three sites) within each station (circles) and generated using the Bootstrap technique with 9999 permutations (squares).

2.5 Discussion

Observations of *Chromis dispilus* as the most abundant species followed by *Scorpis violaceus*, *Chrysiptera rapanui*, *Pseudolabrus luculentus*, *Plagiotremus tapeinosoma* and *Kyphosus bigibbus* are similar to observations by Schiel et al. (1986) and Cole et al. (1992). *Scorpis violaceus* was classified by Francis et al. (1987) as abundant and observed by Cole et al. (1992) at two of four sites surveyed, but was not observed by Schiel et al. (1986). In this study, *Scorpis violaceus* was observed at one of three sites; Denham 2. The high abundance of *Plagiotremus tapeinosoma* noted in my study was not reported by Schiel et al. (1986) nor by Cole et al. (1992) who recorded it as the 10th most abundant benthic fish out of 12, although Francis et al. (1987) reported it to be common which may indicate that this species is only found at specific locations throughout the archipelago or that it varies in its distribution temporally.

Planktivores dominated trophic group abundance at all sites and accounted for almost half of biomass, while invertebrate feeders, herbivores, fish and invertebrate feeders and omnivores represented smaller proportions, suggesting that plankton is a major food source available to fishes of the Kermadec Islands. Omnivores accounted for a high biomass at M, in comparison to small biomasses at both D1 and D2, also suggesting that they are sensitive to differences among sites. Differences in size-class distributions may suggest that certain regions of the Kermadec Islands are preferentially used by juveniles and adults of some species, although species-specific factors have not been identified. These opportunistic observations from the Kermadec Islands provide insight about the trophic structure of reef fishes from these isolated, subtropical waters. Future observations will undoubtedly provide a greater understanding of spatial and temporal variability in reef fish assemblages at the Kermadec Islands MR.

Based on estimates of population sizes and densities of marine communities at the Kermadec Islands, it has been suggested that recruitment may be a limiting factor in a region that is highly isolated, subject to strong wave and storm energy manifested through cyclones as well as the relatively small amount of suitable rocky-reef habitat available due to water depths dropping to as deep as 3000 m within the marine reserve (Schiel et al. 1986). It has also been noted that there are no major currents flowing to the Kermadec Islands, which may explain why major coral and macroalgal groups that are expected to be able to survive in such conditions are absent (Schiel et al. 1986). The Kermadec MR protects a unique marine community that appears to be extremely susceptible to human influence. A major reason why the Kermadec Islands are still home to spotted black grouper, Galapagos reef sharks, endemic giant limpet and Kermadec scalyfin is the geographical isolation of the islands, which were for a large part of history not regular fishing grounds. The MR that is in place today protects a pristine marine ecosystem not found elsewhere in the world.

Chapter 3

Response of Reef Fishes to Marine Reserve Protection Among Sites of Varying Habitat Quality at Kapiti Marine Reserve, New Zealand

3.1 Abstract

Kapiti MR, located in the Cook Strait region, is New Zealand's fourth oldest MR, established in 1992, however one of the least studied. I monitored reef fishes at protected and unprotected sites using underwater observation and compared results with previous studies to determine time scales for direct (response of species targeted by humans) and indirect (response of species not targeted by humans) effects. Analysis of monitoring data has highlighted the importance of baseline studies, as one of the protected sites showed a high abundance of reef fishes prior to MR implementation. Targeted species of reef fishes were shown to increase between 300 – 400% at sites protected by the Kapiti MR. Results indicate that timelines are variable and species-specific for the response of size, abundance and biomass of targeted and non-targeted reef fishes to MR protection.

3.2 Introduction

Many studies have reported the impacts of marine reserve establishment in temperate areas of the world (Bell 1983, McCormick and Choat 1987, Buxton and Smale 1989, Garcia-Rubies and Zabala 1990, Bennett and Attwood 1991, Dufour et al. 1995, Edgar and Barrett 1997). In New Zealand, MR protection has led to increased size and abundance of several commercially and recreationally important fish and shellfish species (MacDiarmid and Breen 1993, Kelly et al. 2000, Willis et al. 2000, Davidson et al. 2002, Willis et al. 2003, Shears et al. 2006). Blue cod (*Parapercis colias*) have been found to be larger inside vs. outside reserves in 9 of 10 studies and more abundant inside reserves vs. outside 8 out of 11 studies at locations throughout New Zealand (Pande et al. 2008). At Poor Knights MR, snapper (*Pagrus auratus*) were found to be more abundant inside the reserve after implementation with an increase in biomass of more than 800%, which was attributed to immigration of adults into the reserve rather than from within-reserve recruitment (Denny et al. 2004). Not all temperate MR studies have documented positive responses for targeted species and it has been suggested that mixed responses occur in species or communities (Cole 1994, Willis et al. 2000) or that sampling methodology is inadequate to detect changes (Kelly et al. 2000).

Langlois and Ballantine describe first, second, third and four order ecological changes that occur after MR protection is employed (2005). First-order changes refer to an increase in exploited species such as snapper and rock lobster, even in relatively small reserves. Second-order changes occur when an increase in predators such as rock lobster cause a decrease in their prey items (urchins) inside of the reserves. Third-order changes are described where densities of herbivores such as urchins decline due to predation allowing kelp beds to regenerate in areas formerly dominated by coralline algae in the urchin barrens. Fourth-order changes occur when areas previously dominated by coralline algae are replaced by kelp beds, resulting in an increase in biodiversity and productivity. This type of trophic cascade has been observed by Shears and Babcock at two MRs in north eastern New Zealand where urchin (*Evechinus chloroticus*) barrens have been controlled through top-down predation by protected lobster (*Jasus edwadsii*) and fish predators (2002).

The Kapiti MR is fourth oldest of the 34 MRs found in New Zealand, established in 1992. Kapiti Island is a popular recreational fishing and diving destination for tourists and residents of the greater Wellington region. As it is located approximately 5km offshore from the mainland, a boat is required to access fishing and dive sites. Given the high human population density on the adjacent mainland, the large proportion of people with boats and often-favourable weather conditions in comparison to the south coast of Wellington, recreational fishing at Kapiti Island accounts for the majority of coastal resource use. While not quantified by the Ministry of Fisheries, recreational fishing effort at Kapiti is substantial, with many boats often 'fishing the line' with hook-and-line at the northern boundary of the MR on the west side of the island (*personal observation*) targeting species such as blue cod (*Parapercis colias*). Spear-fishers are also regularly observed in the waters surrounding the island, targeting benthic-associated reef fishes such as butterfish (*Odax pullus*) as well as larger pelagic species such as kingfish (*Seriola lalandi*) and each year Kapiti Island plays host to the North Island spear-fishing competition. Gill-netting and long-lining by recreational fishers are also permitted in New Zealand, targeting species such as moki (*Latridopsis ciliaris*; *Cheilodactylus spectabilis*) and snapper (*Pagrus auratus*) respectively. Given this exploitation and the responses of targeted reef fishes to protection at other MRs in New Zealand, it is expected that Kapiti MR, now 18 years old, supports a higher abundance, biomass and larger species of targeted reef fishes.

A baseline study was conducted in 1992 that surveyed 11 sites on the north, east and western sides of Kapiti Island (Battershill et al. 1993). While this was a one–off study with only six replicate transects of reef-fish size and abundance undertaken at each site, it identified varying habitat quality among sites located inside the MR (Battershill et al. 1993). Pande, a previous PhD student at Victoria University of Wellington conducted part of her PhD work at Kapiti MR (2001) with monitoring surveys at four of the original 11 sites surveyed by Battershill et al. (1993) from 1998 – 2001. Pande's survey effort was much greater than the baseline survey, however since the baseline survey had a very low level of replication and a large amount of variation, no statistically significant changes in size or abundance of reef fish between surveys were observed, however blue moki and butterfish were found to be larger inside the MR in comparison to outside (2001). Surveys conducted by Stewart and MacDiarmid in 1999 and 2000 (2003) found statistically significant greater sized blue cod and butterfish inside the MR in comparison to outside sites. Struthers, a former MSc student at Victoria University of Wellington used three different survey techniques to monitor blue cod at Kapiti MR and found that there are more and larger blue cod inside the MR in comparison to outside, however underwater visual census (UVC) had the least power to detect statistically significant changes (2004).

To date, the only differences in size and abundance of reef fishes that have been reported at Kapiti MR are inside-outside comparisons (Pande 2001; Stewart and MacDiarmid 2003; Struthers 2004). None of the studies that made comparisons to the baseline study showed any statistically significant results which is likely due to the low replication and high variability in the baseline dataset (Battershill et al. 1993; Pande 2001; Stewart and MacDiarmid 2003; Struthers 2004). Also, one of the inside sites (Arapawaiti) supported a higher abundance of reef fishes prior to MR implementation (Battershill et al. 1993) which makes inside/outside comparisons problematic for detecting MR effects. For these reasons, I have focused on individual sites throughout time instead of pooling inside and outside data and have only conducted statistical tests between my observations and the study by Pande; which represents the most comprehensive study to date (2001).

3.3 Methods

3.3.1 Study Site

Kapiti Island is located in the northwestern Cook Strait region of New Zealand, approximately 50 km north of Wellington and 5 km offshore (40° 51’S, 174° 55’E; Figure 3.1). This region is a convergence zone for the cold Southland current and the warm d’Urville current. Kapiti MR protects an area of 2167 ha, divided into two areas; 1825 ha on the eastern side and 342 ha on the western side of the island (Figure 3.1). This study is focused on the western side of the island, which is exposed to high wave and wind energy from the Tasman Sea as well as strong currents. Subtidal habitat at study sites is characterized by rocky reef platform that slopes off to sand and gravel substrates at approximately 20m.

3.3.2 Underwater Observations

Underwater observations of reef fish size and abundance were conducted from February 2008 until February 2010. During this time, five underwater surveys were conducted seasonally as weather and logistics permitted. For each survey, all four sites were sampled to determine abundance and size of reef fishes. 18 species were surveyed but due to low abundance, detailed analyses are limited to these 7 species, of which the first four are targeted by fishers; blue cod (*Parapercis colias*), blue moki (*Latridopsis ciliaris*), butterfish (*Odax pullus*), red moki (*Cheilodactylus*

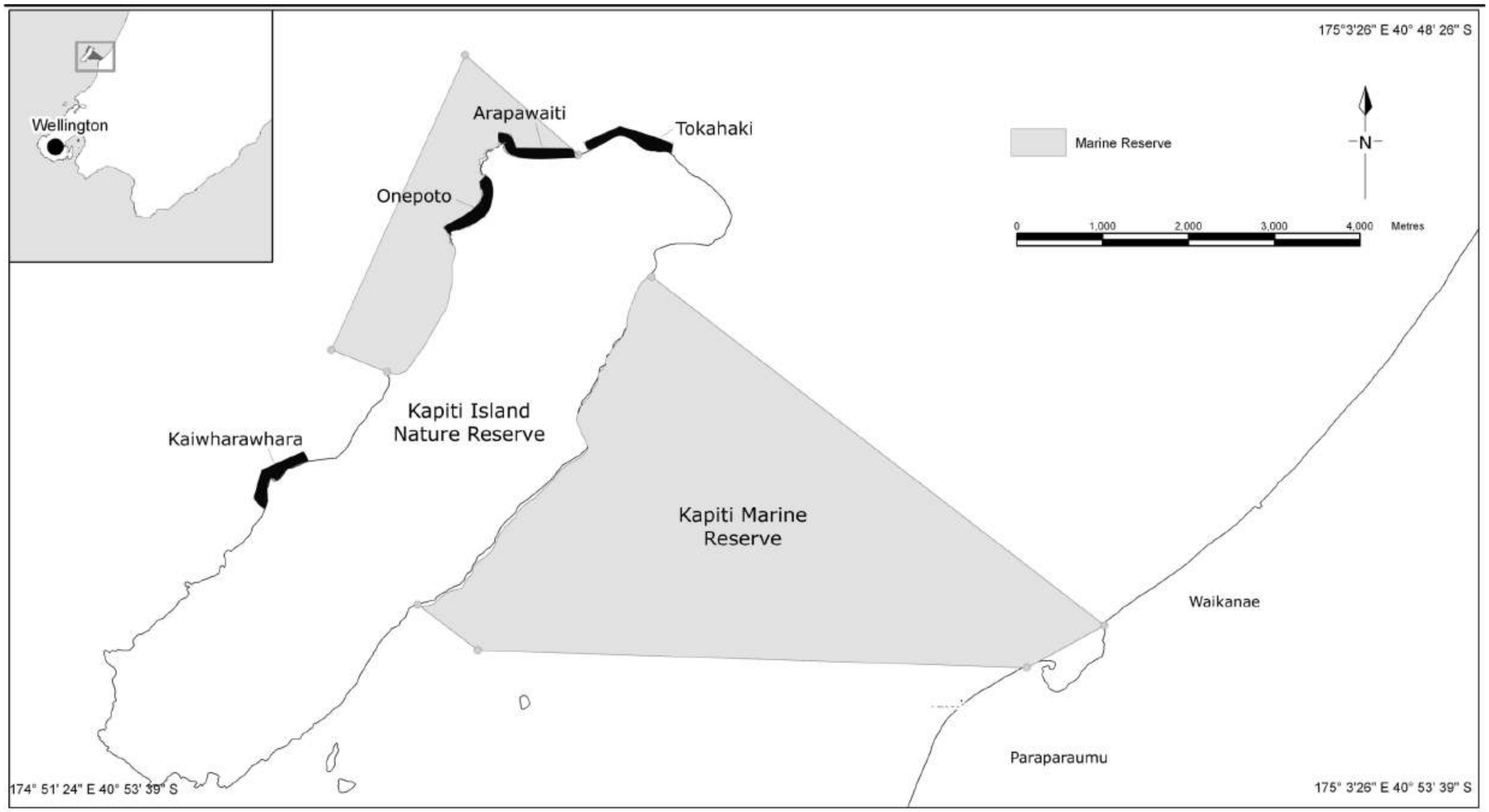


Figure 3. 1. Map of Kapiti Island Marine Reserve showing four study sites (coloured in black). Map used with permission from Pande (2001).

spectabilis), banded wrasse (*Notolabrus fucicola*), scarlet wrasse (*Pseudolabrus miles*) and spotty (*Notolabrus celidotus*). At each site, 9 swimming transects were conducted between 5 and 15m depth. All underwater transects were conducted over rocky reef habitat which was dominated by brown macroalgae. Each transect was started 5m from where the transect tape was set in order to minimise inaccurate counts of fish attracted to or repelled from the disturbance (Cole et al. 1990; Cole 1994). Fish were counted as the transect tape was being deployed rather than retrieved to avoid counting fish attracted to the diver (Cole 1994). Each transect was 5m wide by 25m long, resulting in an area of 125m^2 surveyed for each transect and a total area of 1125m^2 surveyed at each site for each season. Fish size was estimated to the nearest 5cm. All observations were made by the author. Size-frequency data were converted into biomass using non-linear length to weight relationships for north-eastern New Zealand reef fishes as described by Taylor and Willis (1998) and FishBase (Froese and Pauly 2005).

3.3.3 Previous Monitoring Surveys

The baseline survey conducted by Battershill et al. (1993) during the summer of 1992 surveyed a total of 11 sites (including the 4 used by Pande, Struthers and in this study) on both eastern and western sides of Kapiti Island. This survey employed a swimming transect technique 10m wide by 25m long for a total area of 250m^2 surveyed for each transect. Three replicate transects were conducted at each of “shallow” and “deep” depth stratifications to a maximum of 20m, for a total of six transects at each site. Surveys conducted by Pande (2001) and Struthers (2004) employed the same survey methodology at the same sites as described above.

In order to make size, abundance and biomass comparisons between surveys by different researchers, only the four sites surveyed by all researchers were used. Abundance densities were standardised per unit area. As it has been identified that the Arapawaiti site supported higher abundances of reef fishes in comparison to surrounding sites prior to MR implementation (Battershill et al. 1993), I have not pooled the reserve sites together and make only same-site comparisons over time. Pande's surveys (2001) represent the most comprehensive study of reef fishes at Kapiti MR, as she undertook a total of 12 surveys seasonally over three years for a total of 108 transects at each site and determined that season was not a significant factor in explaining variation in size and abundance of reef fishes. Struthers (2003) conducted 4 surveys during all seasons in one year for a total of 36 transects per site. My research conducted 5 surveys over three summers for a total of 45 transects per site. In comparison, the baseline survey (Battershill et al. 1993) conducted only 6 transects at each site for one year and the surveys conducted by Stewart and MacDiarmid (2003) used a total of 16 transects per site over two years. Abundance trends are examined using surveys by Battershill (1993), Pande (2001), Struthers (2003) and this study to give an indication of change throughout MR protection, however statistical comparisons are limited to the study by Pande and this study due to low transect replication in the baseline survey (Battershill et al. 1993). This provides data for the years, 1998 – 2000 and 2008 – 2010, encompassing the last 13 of the 19 years since the Kapiti MR has been in place.

3.3.4 Statistical Analyses

Permutational multivariate analysis of variation (PERMANOVA) was undertaken using PRIMER with PERMANOVA+ software (Anderson et al. 2008). Bray-Curtis similarity matrices were constructed for abundance, size and biomass for all species and for each species. First, an unrestricted one-way PERMANOVA+ test was run using 9999 random permutations and if the test was statistically significant ($\alpha = 0.05$), pairwise tests were conducted to determine if abundance, size and biomass of reef fishes differed between time periods for each site. Permutational analysis of multivariate dispersions (PERMDISP) is analogous to Levene's test to determine if differences exist between the dispersion of groups (Anderson 2004; Levene 1960). PERMDISP was undertaken on Bray-Curtis similarity matrices using PRIMER with PERMANOVA+ software (Anderson et al. 2008) to determine if statistically significant differences between dispersions of reef fish abundance, size and biomass existed between time periods for each site. PERMDISP was used in tandem with PERMANOVA+ to determine if groups that differed also showed differences in dispersions (Anderson 2004). I controlled for false discovery rate (FDR) using the technique described by Verhoeven et al. (2005). As size information was not available for all species surveyed by Pande (2001), size and biomass analyses were only conducted for some species. Observations of reef fishes were pooled to give total abundance, abundance of targeted and non-targeted species, and biomass of targeted species at each site and analysed for differences between periods.

3.4 Results

3.4.1 Abundance trends since MR implementation

Observations of reef fishes at Kapiti Marine Reserve during the 18 years since MR implementation are highly variable within each survey and also show large fluctuations over time (Figure 3.2). The Battershill et al. (1993) baseline survey shows a very high abundance of spotty at Onepoto, Kaiwharawhara and Tokahaki sites with approximately 30 individuals per transect (Figure 3.2B, C, D). At both Onepoto and Kaiwharawhara sites, the high abundances of spotties decreased to approximately 10 per transect by the time of Pande's survey in 1998 (2001), however, Pande also observed high abundances of spotties at Tokahaki. At

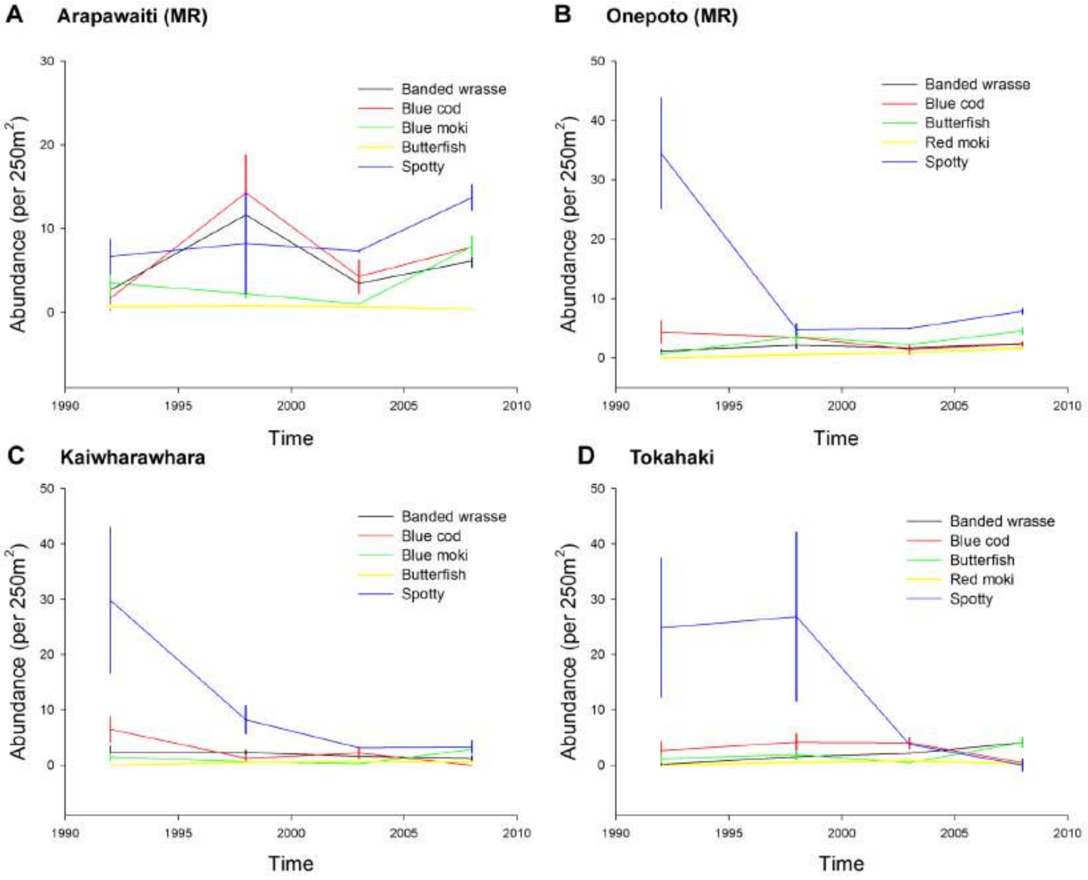


Figure 3. 2. Abundance of reef fishes (with SE) throughout MR implementation at Kapiti Island sites. Panels A & B represent sites protected by the Kapiti MR and panels C & D represent control sites. Data are provided by Battershill et al. 1993, Pande 2001, Struthers 2003 and this study.

Arapawaiti, spotties were at much lower abundance initially at approximately 10 individuals per transect, which increased to approximately 15 by 2008 (Table 3.1).

Blue cod showed both the greatest fluctuation between studies and highest abundances at Arapawaiti, at approximately 10 individuals per transect (Figure 3.2A). At other sites, blue cod fluctuated between studies, however to a lesser degree and at not observed at the same abundance as at Arapawaiti (Figure 3.2). Interestingly, banded wrasse also displays this trend at Arapawaiti, showing the

Table 3. 1. Average abundance, size and biomass of reef fish species per transect (125m^2) with standard error (SE) at sites surveyed during 1998 – 2000 by Pande (upper case) and during 2008 – 2010 by Eddy (lower case). A = Arapawaiti (MR); O = Onepoto (MR); K = Kaiwharawhara (unprotected); T = Tokahaki (unprotected). For the 1998 – 2000 data, n = 12 and for the 2008 – 2010 data, n = 5.

Abundance

	Banded wrasse		Blue cod		Blue moki		Butterfish		Red moki		Scarlet wrasse		Spotty	
Site	$\bar{x}$	SE	$\bar{x}$	SE	$\bar{x}$	SE	$\bar{x}$	SE	$\bar{x}$	SE	$\bar{x}$	SE	$\bar{x}$	SE
A	5.815	0.946	7.139	2.256	0.343	0.147	1.111	0.256	0.315	0.070	3.333	0.776	10.806	2.966
a	3.067	0.778	3.889	0.964	0.222	0.070	3.933	1.217	0.200	0.065	2.356	0.584	6.844	1.523
O	1.139	0.201	0.472	0.154	0.102	0.052	0.241	0.067	0.194	0.057	0.389	0.227	3.537	1.240
o	0.622	0.459	0.000	0.000	0.000	0.000	1.444	0.630	0.289	0.134	0.422	0.108	1.644	1.149
K	0.889	0.246	1.694	0.398	0.639	0.184	1.222	0.607	0.250	0.048	0.833	0.335	1.879	0.542
k	1.222	0.233	1.156	0.323	0.156	0.075	2.267	0.569	0.267	0.083	0.844	0.278	1.422	0.485
T	0.769	0.192	1.731	0.667	0.176	0.055	0.676	0.304	0.157	0.050	0.444	0.161	13.426	7.621
t	2.044	0.653	0.244	0.089	0.067	0.027	2.067	0.877	0.156	0.057	0.600	0.174	3.933	1.082

Size

	Banded wrasse		Blue cod		Blue moki		Butterfish		Red moki		Spotty	
Site	$\bar{x}$	SE	$\bar{x}$	SE	$\bar{x}$	SE	$\bar{x}$	SE	$\bar{x}$	SE	$\bar{x}$	SE
A	23.515	0.748	22.513	0.532	34.441	3.192	31.726	3.167	27.256	2.614		
a	24.479	0.851	25.527	0.622	27.188	5.090	41.942	2.508	44.167	3.727	10.582	0.919
O	20.175	1.411	26.585	1.629	33.795	2.558	27.965	2.525	33.455	3.264		
o	26.928	1.854	32.296	1.578	30.313	3.303	46.218	1.895	47.633	2.126	11.603	1.266
K	20.184	0.980	22.015	1.353	24.958	1.888	24.407	1.495	25.407	4.673		
k	18.674	0.468					40.988	5.989	42.024	3.128	11.006	1.200
T	20.407	1.754	23.933	1.552	17.369	1.264	21.697	2.248	30.488	3.992		
t	23.524	3.331	21.875	0.722	38.333	4.655	34.355	2.097	45.417	7.312	10.574	0.656

Biomass

	Banded wrasse		Blue cod		Blue moki		Butterfish		Red moki		Scarlet wrasse		Spotty	
Site	$\bar{x}$	SE	$\bar{x}$	SE	$\bar{x}$	SE	$\bar{x}$	SE	$\bar{x}$	SE	$\bar{x}$	SE	$\bar{x}$	SE
A	2.092	0.354	1.614	0.508	0.245	0.086	0.646	0.234	0.107	0.027	0.009	0.009		
a	1.033	0.189	1.298	0.336	0.175	0.104	6.402	2.813	0.306	0.113	0.455	0.166	0.261	0.076
O	0.259	0.090	0.621	0.199	0.733	0.335	0.413	0.158	0.208	0.056	0.010	0.007		
o	0.605	0.170	0.766	0.231	0.144	0.101	4.790	1.448	0.505	0.165	0.121	0.028	0.065	0.022
K	0.281	0.088	0.135	0.061	0.063	0.040	0.074	0.028	0.116	0.053	0.001	0.001		
k	0.091	0.067	0	0	0	0	1.808	0.904	0.435	0.214	0.041	0.012	0.047	0.028
T	0.336	0.173	0.443	0.142	0.041	0.018	0.141	0.084	0.141	0.088	0.002	0.001		
t	0.481	0.081	0.054	0.022	0.081	0.046	2.030	1.341	0.237	0.114	0.102	0.027	0.150	0.042

same pattern of abundance and fluctuations throughout time (Figure 3.2A). At other sites, banded wrasse was observed in lower abundance than at Arapawaiti, however once again at similar abundances as blue cod (Figure 3.2B, C, D).

Butterfish showed an increase in abundance from my study at all sites, after showing relatively stable dynamics during previous studies (Figure 3.2). The highest abundance was observed at Arapawaiti, with approximately 8 individuals per transect in comparison to approximately 4 at other sites (Table 3.1; Figure 3.2). Red moki was observed to remain relatively stable throughout time with low abundances of less than one individual observed per transect at all sites (Figure 3.2).

3.4.2 Statistical comparisons from 1998 – 2010

A PERMANOVA test of significance indicated that season was not a significant factor in explaining variation in abundances of reef fishes during my study from 2008 – 2010 (Psuedo F = 0.644; P = 0.718; U = 9574). This is consistent with previous observations at Kapiti MR (Pande and Gardner, *In press*). Results from PERMANOVA pairwise tests between the study conducted by Pande (2001) and this study for each site indicate no significant changes in reef fish abundance were observed after correcting for FDR (Table 3.2). Species that showed differences in abundance dispersions between studies were blue cod, blue moki and scarlet wrasse at the unprotected site Kaiwharawhara (Tables 3.2 & 3.3).

Pairwise tests for average size of reef fishes between studies showed significant differences for butterfish at Onepoto and Kaiwharawhara (Table 3.2). At the protected site Onepoto, average size increased from 28.0 (+/- 2.5cm SE) to 46.2cm (+/- 1.9cm SE), indicating an increase of 18.2cm (Table 3.1). At the unprotected site Kaiwharawhara, average size increased from 24.4 (+/- 1.5cm) to 41.0cm (+/- 6.0cm), indicating an

Table 3. 2. Results of PERMANOVA pairwise tests between time periods for each site. Abundance, size and biomass of reef fish species per transect (125m^2) at each site surveyed during 1998 – 2000 by Pande were compared to surveys conducted in 2008 – 2010 by Eddy. A = Arapawaiti (MR); O = Onepoto (MR); K = Kaiwharawhara (unprotected); T = Tokahaki (unprotected). For the 1998 – 2000 data, n = 12 and for the 2008 – 2010 data, n = 5. t = test statistic; P = permutation p-value; U = number of unique permutations.

Abundance

	Banded wrasse			Blue cod			Blue moki			Butterfish			Scarlet wrasse			Spotty		
Site	t	P	U	t	P	U	t	P	U	t	P	U	t	P	U	t	P	U
A	1.422	0.140	4939	0.680	0.676	4973	0.649	0.673	250	1.824	0.040	2628	0.979	0.371	1627	0.531	0.827	4933
O	0.942	0.411	1625	0.542	0.704	1904	1.389	0.124	611	1.531	0.111	507	0.876	0.473	1494	0.725	0.662	2070
K	1.735	0.038	2091	2.838	0.010	149	1.410	0.267	16	1.249	0.170	860	1.600	0.084	342	1.586	0.072	1636
T	1.757	0.062	1633	1.211	0.225	730	0.771	0.639	56	1.957	0.056	448	1.098	0.376	702	0.840	0.534	3323

Size

	Banded wrasse			Blue cod			Blue moki			Butterfish			Red moki		
Site	t	P	U	t	P	U	t	P	U	t	P	U	t	P	U
A	0.784	0.445	4941	3.155	0.006*	4916	1.198	0.242	431	2.173	0.039	2633	2.900	0.015*	606
O	2.638	0.021	2893	1.862	0.069	4932	0.606	0.567	421	3.635	0.008*	791	2.135	0.039	1468
K	0.791	0.449	858							3.142	0.004*	1990	1.552	0.104	375
T	0.908	0.383	3320	0.611	0.565	780				2.273	0.010	126	1.479	0.158	330

Biomass

	Banded wrasse			Blue cod			Butterfish			Spotty		
Site	t	P	U	t	P	U	t	P	U	t	P	U
A	1.550	0.111	4947	0.268	0.980	4896	2.275	0.003*	3302	0.371	1.000	21
O	1.321	0.153	2220	0.956	0.378	4052	1.870	0.025	792	1.606	0.143	21
K	1.653	0.073	1447				1.767	0.031	1130	0.489	1.000	6
T	2.265	0.015	4122	1.782	0.0372	779	1.050	0.353	126	2.625	0.162	6

* Indicates statistically significant result ($\alpha = 0.05$) using FDR approach.

increase of 16.6cm (Table 3.1). Red moki showed a statistically significant increase of 16.9cm in average size, from 27.3 (+/- 2.6 cm) to 44.2cm (+/- 3.7 cm) at the protected site Arapawaiti (Tables 3.1 & 3.2). Blue cod showed a statistically significant increase in average size at Arapawaiti; increasing from 22.5 (+/- 0.5 cm) to 25.5 cm (+/- 0.6cm), a change of 3cm (Tables 3.1 & 3.2). Pairwise tests for differences in size distributions of reef fishes did not show any significant differences (Table 3.3).

Butterfish showed statistically significant changes in biomass at the protected site, Arapawaiti (Table 3.2). Average biomass increased from 0.6 (+/- 0.2 kg) to 6.4kg (+/- 2.8 kg), indicating a 10-fold increase (Tables 3.1 & 3.2; Figure 3.3). Pairwise tests for biomass distributions indicate that banded wrasse differed at Tokahaki between studies (Table 3.3).

Pairwise tests for differences in total abundance, total abundance of targeted species, and total abundance of non-targeted species between time periods at each site did not show any significant differences. A pairwise test for biomass of targeted species between periods at each site indicated statistically significant increases at both MR sites and also at Kaiwharawhara (Table 3.4). Biomass increased from 2.6 to 8.2 and 2.0 to 6.2 kg per transect at Arapawaiti and Onepoto respectively, indicating approximately 3-fold increases in biomass (Table 3.4). At Kaiwharawhara, biomass of targeted species was observed to be very low by Pande (2001) with an average of 0.39kg per transect in comparison to 2.2kg per transect in this study (Table 3.4). Biomass of targeted species at Tokahaki also increased; from 0.8 to 2.4kg per transect, however this increase was not statistically significant (Table 3.4).

Table 3. 3. Results of PERMDISP pairwise tests between time periods for each site. Dispersions of abundance, size and biomass of reef fish species per transect (125m^2) at each site surveyed during 1998 – 2000 by Pande were compared to surveys conducted in 2008 – 2010 by Eddy. A = Arapawaiti (MR); O = Onepoto (MR); K = Kaiwharawhara (unprotected); T = Tokahaki. For the 1998 – 2000 data, n = 12 and for the 2008 – 2010 data, n = 5. t = test statistic; P = permutation p-value.

Abundance

	Banded wrasse		Blue cod		Blue moki		Butterfish		Red moki		Scarlet wrasse		Spotty	
Site	t	P	t	P	t	P	t	P	t	P	t	P	t	P
A	0.394	0.772	0.356	0.791	0.914	0.544	0.527	0.673	0.042	0.971	3.144	0.027	0.679	0.633
O	2.235	0.146	0.654	0.576	0.690	0.676	3.959	0.011	0.402	0.862	2.444	0.089	1.198	0.365
K	2.442	0.043	8.655	0.000*	5.328	0.001*	0.185	0.860	0.123	0.934	6.445	0.001*	1.120	0.404
T	0.437	0.732	1.044	0.451	0.897	0.346	1.921	0.117	1.866	0.154	3.153	0.034	2.139	0.133

Size

	Banded wrasse		Blue cod		Blue moki		Butterfish		Red moki	
Site	t	P	t	P	t	P	t	P	t	P
A	0.734	0.505	1.117	0.286	1.507	0.224	1.564	0.179	1.538	0.173
O	1.720	0.207	1.527	0.233	0.216	0.836	1.809	0.160	2.372	0.088
K	2.481	0.026					0.920	0.438	1.755	0.202
T	0.623	0.605	1.740	0.148			2.033	0.032	0.439	0.692

Biomass

	Banded wrasse		Blue cod		Butterfish		Spotty	
Site	t	P	t	P	t	P	t	P
A	1.000	0.546	0.400	0.777	0.078	0.950	0.966	0.563
O	1.299	0.381	0.578	0.671	0.606	0.694	1.543	0.534
K	0.495	0.692			0.617	0.605	2.391	0.168
T	3.929	0.002*	1.151	0.454	0.519	0.640	2.226	0.172

* Indicates statistically significant result ($\alpha = 0.05$) using FDR approach.

3.5 Discussion

The large amount of variation in abundance of reef fishes at Kapiti Island highlights the importance of having highly replicated studies in order to be able to detect statistically significant effects of MR protection. One-off studies are of limited use, unless they employ a high sampling effort during that 'snapshot' study. This is the first study that has been able to detect statistically significant changes in size and biomass of reef fishes at Kapiti MR sites throughout time and it is likely due to the fact that all of the previous studies made comparisons to the baseline study. By using the extensive study by Pande (2001) and this study, the power to detect temporal changes is greater. The result that season is not a significant factor in explaining variation in abundance of reef fishes for sites inside and outside the Kapiti MR is similar to the result found by Pande (2001; Pande and Gardner *In press*) however differs from observations at the nearby Taputeranga MR (located approximately 50km away; Pande 2001). This may be due to the difference in oceanographic conditions between the two sites as they are subjected to differing magnitudes of wave, wind and current energy, with Kapiti Island having less variation among seasons.

My findings that size and/or biomass of blue cod, butterfish and red moki have increased over the last 10 years of the 18 years the Kapiti MR has been implemented indicates that direct effects (recoveries of targeted species) of biomass and mean size may take place on decadal time scales. Statistically significant changes in the abundance of targeted species were not observed in the last 10 of the 18 years of MR implementation, however observations by Babcock et al. (2010) at MRs in the temperate waters of New Zealand, Australia and California indicated that direct effects of abundance often occurred within the first decade. Presumably this has also occurred at the Kapiti MR and was not detected statistically, due to the large variation in abundance in the baseline study (Battershill et al. 1993; Pande 2001). Models of reef

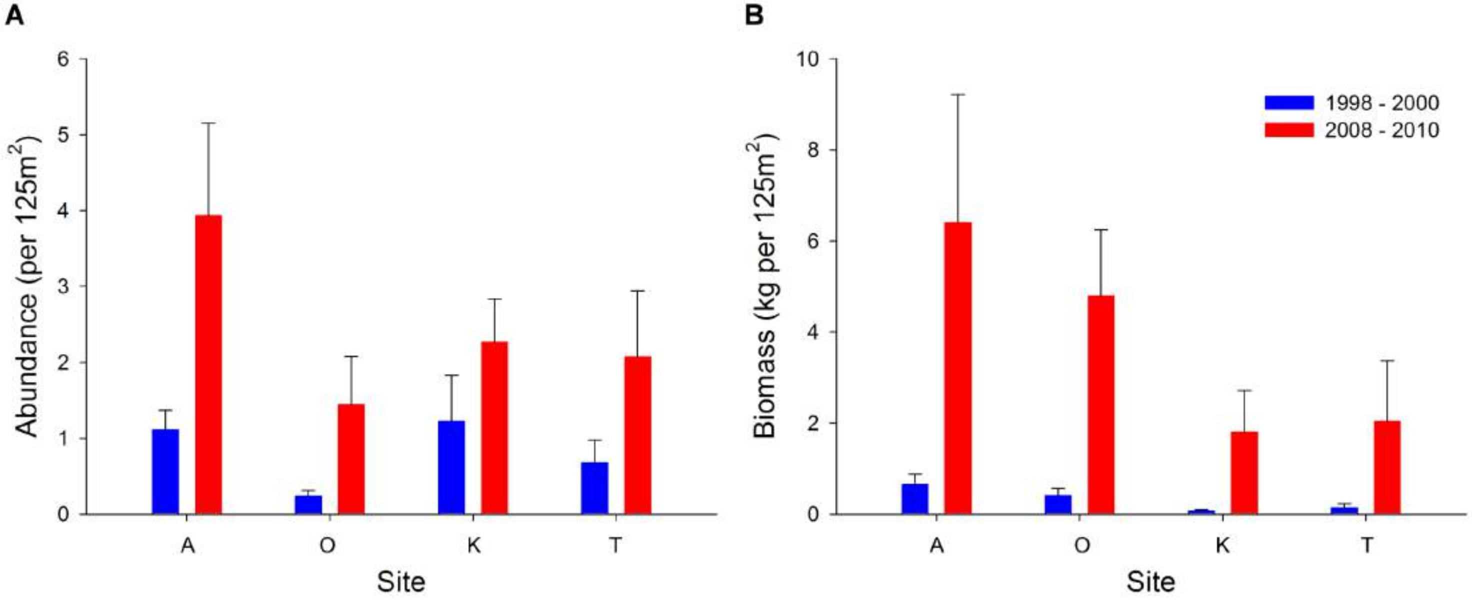


Figure 3. 3. Abundance (panel A) and biomass (panel B) of butterfish (with SE) at Kapiti Island sites for the 1998 – 2000 survey (Pande 2001) and the 2008 – 2010 survey (this study). A = Arapawaiti (MR), O = Onepoto (MR); K = Kaiwharawhara (control); T = Tokahaki (control).

fish response to MR implementation have shown that biomass is slower to recover than abundance as abundance responds with a logistic curve and biomass with an exponential cure (Stockwell et al. 2009). It is likely that the abundance of reef fishes protected in the MR is now stable as evidenced by the relatively small changes in abundance between my study and Pande's study, however as individual fish grow larger, biomass should continue to increase which is supported by my results. The large amount of annual variation in abundance that was observed in this and all previous studies is likely to continue which maybe influenced by strong year classes of different species becoming established. It does not appear that density dependence in settlement or survival is in operation for butterfish due to the large amount of un-grazed kelp available at all sites surveyed.

For his Masters thesis, Struthers compared three techniques to survey abundance of blue cod at Kapiti MR (2003). Blue cod is a species that has been shown to respond positively to MR protection in New Zealand as it is a highly targeted species with relatively low movement rate (Struthers 2003; Pande et al. 2008). Struthers was able to detect statistically significant inside/outside differences using a baited underwater video (BUV) technique (2003). It was noted that this technique is more financially intensive than UVC and it is also limited to surveying only those reef fishes that are attracted to the bait. Increase in butterfish biomass would not have been observed using the BUV technique as this species is a herbivore. Also, given the previously identified high abundance of reef fishes at Arapawaiti before MR implementation, the inside/outside comparison does not tease out the reserve effect.

Arapawaiti was identified by fishers prior to MR implementation as a good fishing location and there was some resistance to closing it to fishing (Pande 2001). In the absence of a baseline survey, fishers' ecological knowledge (FEK) can be important in identifying areas of high habitat quality or areas of high abundance of reef fishes (Johannes et al. 2001). The Arapawaiti site which provides habitat that supports a

Table 3. 4. Average biomass of targeted reef fishes per transect (kg per $125m^2$) with standard error (SE) at sites surveyed during 1998 – 2000 by Pande (upper case) and during 2008 – 2010 by Eddy (lower case). A = Arapawaiti (MR); O = Onepoto (MR); K = Kaiwharawhara (unprotected); T = Tokahaki. Pairwise tests between time periods for each site (P = PERMANOVA test; D = PERMDISP test). For the 1998 – 2000 data, n = 12 and for the 2008 – 2010 data, n = 5. t = test statistic; P = permutation p-value; U = number of unique permutations.

Site	$\bar{x}$	SE	Site (test)	t	P	U
A	2.612	0.566	A (P)	2.384	0.018*	4936
a	8.181	2.595	A (D)	0.072	0.950	
O	1.975	0.339	O (P)	2.598	0.007*	4973
o	6.206	1.451	O (D)	0.239	0.862	
K	0.389	0.084	K (P)	2.832	0.005*	4947
k	2.243	0.811	K (D)	0.684	0.548	
T	0.766	0.168	T (P)	1.066	0.324	4958
t	2.402	1.355	T (D)	0.766	0.568	

* Indicates statistically significant result (α = 0.05).

higher abundance of reef fishes is an example why it is important to conduct baseline surveys before MRs are implemented. Habitat structure was identified as being different at Arapawaiti than the other sites monitored in that it was composed of large *Ecklonia radiata* stands as well as boulder barrens (Battershill et al. 1993; Pande 2001). Detecting the effects of MR implementation on size, abundance and biomass of reef fishes is problematic when inside/outside comparisons are confounded by the effect of habitat quality. The technique I have employed to address this issue is to use only same-site comparisons among surveys. While this technique also may pose issues if observers and survey methods have changed over time, the effect of habitat quality is not a confounding factor. However, if habitat quality changes over time, it may influence assemblages of reef fishes and confound the results.

At Arapawaiti, the tenfold increase in biomass of butterfish likely has an impact on the biomass of kelp as adult butterfish exclusively graze on brown algal species and have been shown to play a major role in the structure of kelp forests in New Zealand (Taylor and Schiel 2010). This phenomenon is largely influenced by wave energy, whereby sites exposed to high wave energy are less impacted by butterfish grazing (Taylor and Schiel 2010). Give the moderate wave exposure at Arapawaiti, the effects of grazing on macroalgae are suggested to have a substantial effect on macroalgae community structure. Reports from grazing on *Eckonia radiata* at Arapawaiti differ between the baseline study (Battershill et al. 1993) where *Ecklonia* were reported to have no damage, and Pande's study, where there was significant damage recorded (2001). This is likely due to the increased abundance and biomass of grazers such as butterfish on *Ecklonia radiata*.

At Arapawaiti, statistically significant increases in the average size of red moki, blue cod and butterfish also were observed during the time between the study by Pande (2001) and this study. Reasons explaining the statistically significant increases at Arapawaiti and not Onepoto, the other reserve site, are likely due to higher abundance and biomass, which therefore increased power to detect change at Arapawaiti. It may also be the case that targeted reef fishes at areas of high habitat quality respond disproportionately to MR protection in comparison to targeted reef fishes at other areas. The observation that butterfish also showed statistically significant increase in size at Kaiwharawhara can be attributed to different explanations. Firstly, it could be that butterfish area 'spilling over' from the MR into neighboring areas (Kellner et al. 2007). Secondly, it maybe the timing of the surveys, whereby a survey during a recruitment event would favour a smaller average size, whereas a survey occurring during a spawning aggregation would likely favour a larger average size. Thirdly, it maybe due to factors associated with the site, as blue cod, blue moki and scarlet wrasse were found to vary significantly among studies at Kaiwharawhara, which was not observed at any other sites.

The statistically significant observation of a threefold increase in biomass of targeted species of reef fishes at both sites protected by the MR indicates that the Kapiti MR is rebuilding exploited fish stocks. This contrasts with statistically significant decreases in biomass of blue cod at Tokahaki, a site favoured by recreational fishers whom are often observed 'fishing the line' (personal observation). The biomass of targeted species of reef fishes at both sites protected by the MR is approximately three times greater than at both unprotected sites. In summary, this study provides the first evidence of a statistically significant increase of reef fish size and biomass increase at Kapiti MR. Previously, butterfish had only been observed to increase at the Cape Rodney – Okakari Point (Leigh or Goat Island) MR, located in the north of New Zealand (Haggit et al. 2008). This study is the first to report butterfish recovery in a MR in central or southern New Zealand, where they are the dominant herbivores (Taylor and Schiel 2010).

The recovery of targeted species of reef fishes to MR protection is important to understanding trophic cascades and unexploited ecosystem structure of the Kapiti Island marine community. These results are consistent with models of biomass recovery for direct effects of MR implementation on reef fishes (Stockwell et al. 2009) and provide additional understanding for temperate MR timelines of recovery (Babcock et al. 2010).

Chapter 4

Influence of Coastal Resource Use on Past, Present and Future Ecosystem States of the Taputeranga Marine Reserve, New Zealand

4.1 Abstract

The Taputeranga MR, located on the south coast of the capital city of Wellington, New Zealand was implemented in 2008 and protects 854 ha. The location of the Taputerana MR in the Cook Strait means that the marine environment it protects is subject to high wind, wave and current energies. Commercial, recreational and traditional fisheries are important in this region and have been documented over the last 70 years and it has been estimated that lobster biomass is now approximately one quarter of its former state. I have used an ecosystem-based modeling approach to analyse the food web linkages on the Wellington south coast immediately before the Taputeranga MR was established for comparison to historic and future ecosystem states. Construction of an Ecopath with Ecosim (EwE) model involved collection of biomass data for all species of marine organisms found on the Wellington south coast from a variety of sources. This involved my own underwater observations of reef fishes, which is a part of the ongoing Taputeranga MR monitoring program. My results suggest that the role of lobster within the ecosystem has changed from historic times as it previously had a much greater keystone role (organising of other species) within the ecosystem than it currently experiences and its diet is predicted to have been more herbivorous in historic times. My results indicate that the Taputeranga MR is capable of restoring the future ecosystem to a state more similar to that observed during historic times.

4.2 Introduction

Given the extent of worldwide fishing pressure on individual marine species as well as entire marine ecosystems, studies that have compared current ecosystem states to historic or pristine states have invariably found that large scale changes have occurred (Eddy et al. 2010; Coll et al. 2008; Rosenberg et al. 2005; Parsons et al. 2009). Traditional fishery management practices have mostly focused on single-species approaches in order to make stock assessments to determine the maximum sustainable yield (MSY) that can be harvested. The ecosystem-based management (EBM) approach has been gaining momentum in its use by government and managing authorities following a widespread call from the academic community for its implementation (Browman et al. 2004). Concurrent with the increase in the amount of literature produced about EBM in the past decade has been the number of definitions of EBM.

> “Although there are a bewildering number of different definitions…” there is a “…widespread agreement about the need to move towards a new fishery management system that recognises explicitly how food web linkages and human interventions may affect sustainability in aquatic ecosystems.” (Pitcher et al. 2009b).

In order to understand food web linkages in aquatic ecosystems, it is necessary to quantify such interactions. The “Rebuilding Global Fisheries” article published in Science by a team of world-leading marine conservationists, fishery biologists, marine ecologists and ecosystem modelers (Worm et al. 2009) used Ecopath ecosystem models to assess the impact of fisheries on marine ecosystems worldwide. This is one of many applications of Ecopath ecosystem models following the software’s development during the past quarter of a century (Polovina 1984a, 1984b). Ecopath software uses a mass-balance approach to quantify transfer of biomass within different trophic groups of an ecosystem and how that biomass is passed through the food web (Christensen and Walters 2004). Ecopath with Ecosim (EwE) is an extension of the original software that allows for dynamic simulation and can be used to reconstruct historic ecosystem dynamics as well as for scenario prediction (Christensen and Walters 2004). Using EwE, it is possible to parameterise fisheries to determine the effects of exploitation on the ecosystem.

Recent Ecopath with Ecosim (EwE) applications have been to determine key ecosystem indicators, historical reconstruction of ecosystems and as tools for management decisions (Libralato et al. 2006; Coll et al. 2007; Coll et al. 2008b; Ainsworth et al. 2008; Ainsworth et al. 2008b; Coll et al. 2009; Coll et al. 2009b Ainsworth et al. 2010). Indicators that have been identified as important are: keystone species, transfer efficiency, biomass of groups, mean trophic level of fishery catch and biomass ratios between functional groups (Jordan et al. 2008; Libralato et al. 2010). Historical ecosystem reconstructions have been undertaken for northern British Columbia, Canada (Ainsworth et al. 2008), North-Central Adriatic and South Catalan Seas, Europe (Coll et al. 2007; Coll et al. 2008b; Coll et al. 2009; Coll et al. 2009b). These models have documented large-scale ecosystem-wide changes that have occurred as a result of fishery harvest among other human-mediated disturbances (Coll et al. 2008b; Link et al. 2010), whereas in other areas it has been determined that trophic control has remained constant over the last century (Ainsworth et al. 2008). Many EwE models have been used to predict impacts of management strategies for ecosystems (Ainsworth and Pitcher 2010; Albouy et al. 2010; Ortiz et al. 2009; Ainsworth et al. 2008b; Okey et al. 2004)

In New Zealand, two static ecosystem models have been published using Ecopath software. Bradford-Grieve et al. (2003) constructed a model for the offshore Southern Plateau Subantarctic region (47° – 55° S, 167° E – 177° W). This model employed 19 trophic groups and determined that the system is characterised by low phytoplankton biomass as primary production is iron-limited. Despite low biomass, the system supports high trophic level predators such as marine mammals as well as a commercial fishery through high transfer efficiency of energy. The long food web is dominated by the microbial loop and 69% of biomass remains in the pelagic system.

The second New Zealand model was constructed for a coastal ecosystem at the Te Tapuwae o Rongokako Marine Reserve, which is located north of Gisborne on the central-east coast of the North Island (~ 39 °S). The Tapuwae o Rongokako Marine Reserve was implemented in 1999 and is 2450 ha in size and protects habitats composed of intertidal reef and sand, as well as subtidal reef and mobile substrates. The model was constructed with 22 trophic groups composed of birds, predatory and grazing invertebrates, detritivores, five groups of fish, microphytes, macroalgae, zooplankton, phytoplankton, bacteria and detritus (Pinkerton et al. 2008).

The research described in this chapter is a continuation and extension of the monitoring programme developed by Pande and Gardner (2008) prior to implementation of the Taputeranga MR. This monitoring programme was designed using the before-after-control-impact design (BACI), in order to establish patterns of seasonal, temporal and spatial variation for species of macroalgae, macroinvertebrates and reef fishes. Prior to establishment of the MR, Pande surveyed 8 sites along the south coast of Wellington, three of which are now protected by the Taputeranga MR, three are located to the east of the MR and two located to the west of the MR (Pande 2001). The easternmost site is Barrett Reef and the westernmost is Sinclair Head, spanning approximately 15 km. In order to assess the impact of the MR on targeted and non-targeted reef fishes, surveys have continued after implementation of the Taputeranga MR at the same sites using the same methodology as employed by Pande (2001). This monitoring programme includes recent information for subtidal macroalgae, invertebrates (Byfield, *Thesis in progress*) and reef fishes (this thesis), as well as intertidal macroalgae and invertebrates (Tam, *Thesis in progress*; Jones, *Thesis in progress*).

Maori have inhabited New Zealand for an estimated 800 years, long before European arrival, with a large dietary reliance on coastal marine resources (Leach 2006). There is evidence of lobster (*Jasus edwardsiii*) in middens located on Wellington's south coast, which were harvested by diving, pots and hoop nets (Booth 2008). The commercial lobster fishery on Wellington's south coast was one of the first lobster fisheries in the country (Booth 2008). In the late 1940's, most of the lobster were harvested on rocky inshore areas between depths of 5 and 25m (Booth 2008). During the late 1970's, lobster were fished to depths of 50m (Booth 2008). There is evidence that the average size of lobster is smaller today than in the 1940's (Booth 2008). Commercial fishing of lobster through the use of pots represents the majority of fishery revenue within the study area. There is also a substantial recreational lobster fishery, taken by both potting and diving within the area. The most recent report indicates that this recreational catch makes up approximately 10% of the total allowable catch (TAC) for the region (CRA4) where the Taputeranga MR is located (Ministry of Fisheries 2009b). Other exploited species of shellfish include the New Zealand abalone (paua; *Haliotis australis*, *H. iris*), which are highly revered species as well as some other species of gastropods.

Blue cod (*Parapercis colias*), a species endemic to New Zealand, is the most sought-after finfish species. There are both commercial and recreational fisheries for this species, however the most recent reports indicate that recreational landings far exceed both the total allowable commercial catch (TACC) and commercial landings (Ministry of Fisheries 2009). From the latest report, blue cod is the second most frequently landed species nationally by recreational fishers. Estimates of recreational landings are 30 times greater than commercial landings for the region (BCO2) where the Taputeranga MR is located. Other recreational and commercial finfish fisheries within the model area target butterfish, blue and red moki, tarakihi, trevally, kahawai, warehou, bluenose and barracouta (Ministry of Fisheries 2009).

It has been shown that exploitation of coastal marine resources affects not only the targeted species, but also other species in the ecosystem. A trophic cascade has been observed by Shears and Babcock at two MRs in north eastern New Zealand where urchin (*Evechinus chloroticus*) barrens have been reduced through top-down predation on the urchin by protected lobster (*Jasus edwadsii*) and fish predators (2002). Langlois and Ballantine describe first, second, third and four order ecological changes that occur after MR protection is employed (2005). First-order changes refer to an increase in exploited species such as snapper and rock lobster, even in relatively small reserves. Second-order changes occur when an increase in predators such as the rock lobster cause a decrease in their prey items (urchins) inside of the reserves. Third-order changes are described where densities of herbivores such as urchins decline due to predation allowing kelp beds to regenerate in areas formerly dominated by coralline algae in the urchin barrens. Fourth-order changes occur when areas previously dominated by coralline algae are replaced by kelp beds, resulting in an increase in biodiversity and productivity.

The aims of this research are threefold: 1) To consolidate data from many different sources into an ecosystem model representing the Taputeranga MR at the time of implementation (2008). 2) Using historic accounts, fishery catch records, and stock assessments to reconstruct a past ecosystem model prior to large-scale exploitation. 3) To predict if the Taputeranga MR ecosystem is able to return to its historic state.

4.3 Methods

4.3.1 Building a New Zealand Ecosystem Model

For anyone who is building an ecosystem model for a coastal region of New Zealand, I recommend the extensive report prepared by Lundquist and Pinkerton (2008) for the Te Tapuwae o Ronokako MR in Gisborne, New Zealand. This report provides a comprehensive review of research and information required for parameterisation of an ecosystem model. While it is always preferable to use information obtained from within the model area, the logistics of acquiring such data could fill many lifetimes and theses of research. Lundquist and Pinkerton (2008) provide information about biomass, production and consumption values as well as diet information for New Zealand studies and provide estimations used in other ecosystem models. The comprehensive report by Shears and Babcock (2007) also provides a large amount of information about shallow subtidal macroalgal and invertebrate community structure at locations throughout the country, as well as information about biomass and energy conversions for many species.

4.3.2 Study Area

My study site, the Taputeranga Marine Reserve (41° 20 S, 174° 45 E) protects 854.79 hectares of coastal waters on the south coast of New Zealand's capital, Wellington. The reserve extends from Princess Bay on the eastern boundary to Quarry Bay on the western boundary (Figure 4.1). The reserve has been the result of many years of work by local residents of the south coast, scientists at Victoria University of Wellington and The Royal Forest and Bird Protection Society of New Zealand. Formal application for the reserve was made by the South Coast Marine Reserve Coalition in October 2000 and the MR was gazetted in August 2008.

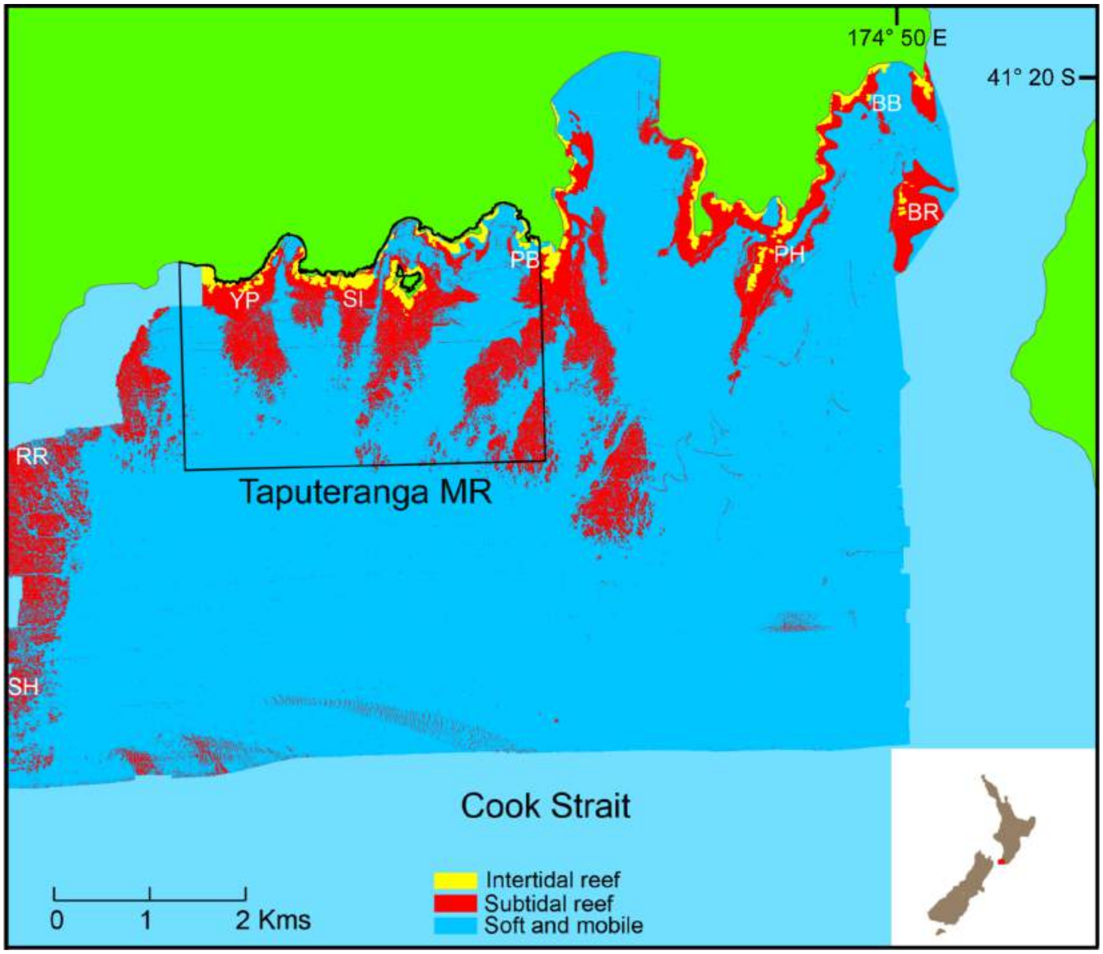


Figure 4. 1. Map of model area (dark blue, red and yellow) with substrate types for ecosystem models showing location of Taputeranga MR. Model area is represented by substrate types; intertidal reef; subtidal reef; and soft and mobile substrates. Study sites for biomass data collection are shown in white letters; BR: Barrett Reef; BB: Breaker Bay; PH: Palmer Head; PB: Princess Bay; SI: Sirens; YP: Yungh Pen; RR: Red Rocks; SH; Sinclair Head. Location of Taputeranga MR within New Zealand as red square in bottom right panel.

The marine environment that the Taputeranga MR protects is representative of the temperate Cook Strait region. This is a highly dynamic area, receiving substantial wave energy from the south as well as the zone of convergence for the East Cape, D'Urville and Southland currents. Habitats protected by the MR include wave exposed rocky reef, wave sheltered rocky reef, cobble beach and sandy shore (Eddy et al. 2008).

Wellington's south coast is home to a diverse assemblage of algal species including macroalgal stands, also known as "kelp forests", which provide habitat for a large number of invertebrate and vertebrate species. These algae belong to the brown algal group (Phaeophyceae), while red (Rhodophyceae) and green (Chlorophyceae) algae are also represented and speciose along the south coast. Much of the bottom type is characterised by greywacke reef, which is structurally complex with caves and gullies that provide habitat for a number of commercially and recreationally targeted invertebrates including lobster (*Jasus edwardsiii*), paua (abalone - *Haliotis iris* and *Haliotis australis*) and kina (urchin - *Evechinus chloroticus*). The combination of macroalgae and rocky reef provides habitat for many fish species typical of Cook Strait temperate assemblages. Encrusting communities are composed of sponges, hydroids, ascidians and bryozoans. Elsewhere in the reserve, the substratum is sand with its associated, but poorly known epifaunal and infaunal assemblages.

4.3.3 Ecological Modeling

Ecopath with Ecosim (EwE) version 6 modeling software was used for the construction of present and historic ecosystem models and for future ecosystem scenario prediction (Christensen and Walters, 2004; www.ecopath.org). EwE uses a mass-balance approach to account for production and consumption of functional groups within the ecosystem (equation 1).

$$B_i \cdot \left(\frac{P}{B}\right)_i = \sum_j B_j \cdot \left(\frac{Q}{B}\right)_j \cdot DC_{ij} + Y_i + E_i + B_i \cdot \left(\frac{P}{B}\right)_i \cdot \left(1 - EE_i\right)$$

(equation 1)

Parameters are described by; B_i = biomass; P/B = production per unit of biomass of the functional group i; $(Q/B)_j$ = consumption per unit of biomass of the predator j of biomass B_j; DC_{ij} = proportion of prey i in the diet of predator j; Y_i = exports from the system as fishery catches; E_i = net migration; EE_i = ecotrophic efficiency of the functional group i. Losses of energy intake for each functional group are represented by equation 2.

$$B_j \cdot \left(\frac{Q}{B}\right)_i = B_j \cdot \left(\frac{P}{B}\right)_j + B_j \cdot \left(\frac{R}{B}\right)_j + \left(\frac{U}{Q}\right)_j$$

(equation 2)

Parameters are described by; $(R/B)_j$ = respiration rate per unit of biomass; $(U/Q)_j$ = fraction of food consumption that is not assimilated.

EwE uses equations 1 and 2 in combination with a predator/prey diet matrix to describe the ecosystem that can be integrated over time to run simulations, represented by equation 3.

$$\frac{dB_i}{dt} = \left(\frac{P}{Q}\right)_i \cdot \sum_{j=1} Q_{ji} - \sum_{j=1} Q_{ij} + I_i - \left(M_i + F_i + e_i\right) \cdot B_i$$

(equation 3)

Described by parameters: dB_i/dt = biomass growth rate of group i during the time interval dt; $(P/Q)_I$ = net growth efficiency; M_i = non-predation natural mortality rate, F_i = fishing mortality rate; e_i = emigration rate; I_i = immigration rate; $I_i - e_iB_i$ = net migration rate. Detailed information about EwE and its strengths and weaknesses have been documented by Christensen and Walters (2004).

EwE analyses provide information about ecosystem structure and function. EwE is able to identify keystone groups which are defined as having low biomass but a structuring role within food webs. EwE employs mixed trophic analysis which is similar to a sensitivity analysis, in that it determines ecosystem-wide impacts of increasing individual groups by small amounts. This analysis can also be used to understand and predict how specific species or trophic groups can cause trophic cascades. The Lindeman spine flow diagram shows transfer efficiency between trophic levels, which has been identified as a key ecosystem indicator (Coll et al. 2009).

4.3.4 Model Parameterisation

I have chosen to use the same 22 trophic groups as Pinkerton et al. (2008), with the addition of two, for a total of 24 (Table 4.1). These trophic groups were chosen based on similarities of morphology, and in the case of consumers, diet composition (Lundquist and Pinkerton 2008). I have added the trophic groups “paua” and “kina” as these animals are observed in relatively high biomass at the Taputeranga MR and are commercially and recreationally important in this region, in contrast to the Te Tapuwae o Ronokako MR which has been observed to be relatively impoverished in terms of grazers (Pinkerton et al. 2008). I parameterised biomasses of trophic groups using observational data collected within the model area for the following trophic groups: lobster, mobile invertebrates - herbivores, paua, kina, mobile invertebrate - carnivores, sea cucumber, sponges, sessile invertebrates, fish – cryptic, fish - invertebrate feeders, fish – piscivores, fish – planktivores, fish – herbivores, macroalgae - canopy, macroalgae – foliose, macroalgae – crustose and phytoplankton. In the absence of data from the model area for all groups, I have used estimations from nearby locations in the Wellington region for the trophic groups phytal/infaunal invertebrates and proportion of fish - invertebrate feeders, fish – piscivores and fish – planktivores groups. For the trophic groups lacking local information - birds, microphytes, meso/macrozooplankton, microzooplankton, bacteria and detritus - biomasses were estimated from values used by Lundquist and Pinkerton (2008). The 'present' ecosystem model refers to the time immediately prior to the implementation of the Taputeranga MR in 2008.

Table 4. 1. Inputs (*B*, *P/B*, *Q/B*) by functional group of the Taputeranga MR ecosystem model representing the historic and present periods. *B*: Initial biomass (gCm^{-2}); L_{rec}: recreational fishery landings; L_{com}: commercial fishery landings; *P/B*: production/biomass ratio (yr^{-1}); *Q/B*: consumption/biomass ratio (yr^{-1}).

	Functional Group	Past	Present				
		B	*B*	L_{rec}	L_{com}	*P/B*	*Q/B*
1	birds	0.0002	0.0002			0.10	89.70
2	lobster	1.64	0.41	0.02	0.18	0.50	7.40
3	mob inverts herb	1.91	0.97			1.30	7.94
4	paua	0.46	0.23	0.15		1.50	15.00
5	kina	0.12	0.06			1.10	7.50
6	mob invert carn	0.61	0.61			1.76	5.97
7	sea cucumber	0.35	0.35			0.60	3.40
8	phytal/infaunal inverts	0.54	0.54			3.67	12.00
9	sponges	1.59	1.59			0.20	0.80
10	sessile inverts	1.56	1.56			1.50	6.00
11	fish cryptic	0.04	0.04			2.40	15.60
12	fish inverts	0.13	0.09			0.41	3.59
13	fish piscivores	0.03	0.01	0.0025		0.43	2.62
14	fish planktivores	0.22	0.15			0.50	6.33
15	fish herbivores	0.37	0.25	0.01	0.08	0.40	9.52
16	microphytes	7.64	7.64			21.00	0.00
17	macroalgae canopy	37.66	37.66			2.87	0.00
18	macroalgae foliose	18.19	18.19			13.00	0.00
19	macroalgae crustose	1.36	1.36			25.40	0.00
20	meso/macrozooplankton	0.17	0.17			17.70	51.50
21	microzooplankton	0.06	0.06			220.00	624.00
22	phytoplankton	0.48	0.48			324.00	0.00
23	bacteria	0.60	0.60			100.00	400.00
24	detritus	1.00	1.00				

Subtidal invertebrate and algal per cent cover and abundance data were collected between 5 and 15 metres depth using a 1 m^2 quadrat placed at 5 m intervals along a 50 m transect at 8 sites within the model area during the summer seasons of two years (2007 – 2009) as a part of Tamsen Byfield's PhD thesis (*In progress*). Collections of macroalgae – canopy (*Cystophora scalaris*, *Carpophyllum flexuosum*, *Carpophyllum maschalocarpum*, *Ecklonia radiata*, *Landsburgia quercifolia*, *Lessonia variegata*, *Macrocystis pyrifera*, *Marginariella bwownii, Marginariella urvillia*, *Sargassum sinclairii*, *Caulerpa brownii*, *Caulerpa flexilis*, *Zonaria turneriana*), paua (*Haliotis iris* and *Haliotis australis*) and kina (*Evechinus chloroticus*) from within the model area were undertaken to determine biomass of individuals (Byfield, *In progress*). Intertidal algal and invertebrate per cent cover and abundance data to species level were collected by randomly placing a 0.5 m by 0.5 m quadrat five times for each of low, middle and high intertidal zones as a part of Jamie Tam and Tim Jones' PhD theses (*In progress*; *In progress*). Six sites were surveyed within the model area during the summer season of (2008 – 2009; Tam, *In progress*; Jones, *In progress*). Conversion of per cent cover and abundance to biomass of macroalgal and invertebrates species that were not collected by Byfield (*In progress*), were determined using ratios from Lundquist and Pinkerton (2008) and Shears and Babcock (2007). Biomass was converted into g.C m^{-2} using ratios for individual species (Lundquist and Pinkerton 2008) and then pooled across trophic groups.

Underwater observations of reef fish size and abundance were conducted seasonally eight times as weather and logistics permitted from August 2007 until February 2010. For each survey, all eight sites were sampled to determine abundance and size of reef fishes. Nineteen species were surveyed but due to low abundance, detailed analyses are limited to 10 species; banded wrasse (*Notolabrus fucicola*), blue cod (*Parapercis colias*), blue moki (*Latridopsis ciliaris*), butterfish (*Odax pullus*), leatherjacket (*Parika scaber*), marblefish (*Aplodactylus arctidens*), red moki

(*Cheilodactylus spectabilis*), scarlet wrasse (*Pseudolabrus milnes*), spotty (*Notolabrus celidotus*) and tarakihi (*Pseudolabrus macropterus*). At each site, 9 transects were surveyed at between 5 and 15 m depth. Each transect was started 5 m from where the transect tape was set in order to minimise inaccurate counts of fish attracted to or repelled from the disturbance (Cole et al. 1990; Cole 1994). Fish were counted as the transect tape was being deployed rather than retrieved to avoid counting fish attracted to the diver (Cole 1994). Each transect was 5 m wide by 25 m long, resulting in an area of 125 m^2 surveyed for each transect and a total area of 1125 m^2 surveyed at each site for each season. Fish size was estimated to the nearest 5 cm. All surveys were conducted by the author. Data were averaged across all seasons. Fish species were assigned to one of four trophic groups; herbivores, planktivores, invertebrate feeders or piscivores (Francis 2001; Froese and Pauly 2005). Size-frequency data were converted into biomass using non-linear length to weight relationships for north-eastern New Zealand reef fishes as described by Taylor and Willis (1998) and FishBase (Froese and Pauly 2005).

Commercial fishery landings data for lobster were obtained from the Ministry of Fisheries May 2009 Plenary Report (Ministry of Fisheries 2009b) to provide information about abundance at depths deeper than those surveyed on SCUBA. Data for the bottom trawl demersal finfish fishery that occurred within 100 km of, and at depths found within the model area were obtained from the Ministry of Fisheries commercial fishers logbook database to provide information about abundance for demersal species (Ministry of Fisheries 2000). Data for pelagic finfish species occurring within 100 km of the model area were obtained from the Ministry of Fisheries aerial sight database (Ministry of Fisheries 2009).

The biomass of the trophic group “cryptic reef fishes” was estimated from an intertidal study that occurred within the model area (Willis and Roberts 1996) and from subtidal observations described by Lundquist and Pinkerton (2008). Size-frequency data were converted into biomass using non-linear length to weight relationships for north-eastern New Zealand reef fishes as described by Taylor and Willis (1998), Lundquist and Pinkerton (2008) and FishBase (Froese and Pauly 2005). Biomass was converted into gC m^{-2} using a ratio of 8.3% carbon to wet weight (Lundquist and Pinkerton 2008).

Phytoplankton biomass was estimated using Chl *a* concentration data from the SeaWifs ocean colour satellite for the period 1997-2006 for an offshore location within the model area (centroid 41° 20 S, 174° 30 E) to minimise impact of coastal runoff (Lundquist and Pinkerton 2008). Phytoplankton production was determined using the vertically generalised production model (VGPM; Behrenfeld & Falkowski 1997).

Phytal invertebrate (organisms living on macroalgae) biomass was estimated as a proportion of macroalgae biomass using methods described by Lundquist and Pinkerton (2008). Infaunal invertebrate biomass was estimated from studies of soft-sediment research that had taken place at Fitzroy Bay which is located ~5 km from the model area and also exposed to high wave energy (Anderlini and Wear 1990).

In the absence of data from the model area, microphyte, meso/macrozooplankton, microzooplanton and bacteria biomasses were estimated from values used by Lundquist and Pinkerton (2008). Detritus biomass was estimated by EwE. Diet, production and consumption values (Tables 1, 2 & 3) were estimated using data from Lundquist and Pinkerton’s extensive report (2008), which reviews a large amount of literature and research in New Zealand and abroad.

Following guidelines laid out by Link (2010), I have employed the pre-balancing routine (PREBAL) in order to ensure that model parameters obeyed energetic laws for ecosystems structure.

Table 4. 2. Diet matrix for present Taputeranga MR ecosystem model. Diets are expressed as a proportion of total diet.

	Prey \ predator	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	20	21	23
1	birds	-	-	-	-	-	$1*10^{-6}$	-	-	-	-	-	-	-	-	-	-	-	-
2	lobster	-	-	-	-	-	$4*10^{-4}$	-	-	-	-	-	-	-	-	-	-	-	-
3	mob inverts herb	0.20	0.21	-	-	-	0.15	-	-	-	-	-	0.12	-	-	-	-	-	-
4	paua	0.03	0.01	-	-	-	0.04	-	-	-	-	-	0.02	-	-	-	-	-	-
5	kina	0.01	0.01	-	-	-	0.01	-	-	-	-	-	0.01	-	-	-	-	-	-
6	mob invert carn	0.29	0.15	-	-	-	0.14	-	-	-	-	-	0.31	-	-	-	-	-	-
7	sea cucumber	-	-	-	-	-	0.06	-	-	-	-	-	-	-	-	-	-	-	-
8	phytal/infaunal inverts	0.31	0.32	-	-	-	0.10	-	-	-	-	0.58	0.17	-	0.05	-	-	-	-
9	sponges	-	-	-	-	-	0.07	-	-	-	-	-	0.04	-	-	-	-	-	-
10	sessile inverts	-	-	-	0.05	0.05	0.43	-	-	-	-	0.24	0.33	-	-	-	-	-	-
11	fish cryptic	0.16	-	-	-	-	-	-	-	-	-	-	-	0.09	-	-	-	-	-
12	fish inverts	-	-	-	-	-	-	-	-	-	-	-	-	0.21	-	-	-	-	-
13	fish piscivores	-	-	-	-	-	-	-	-	-	-	-	-	0.09	-	-	-	-	-
14	fish planktivores	-	-	-	-	-	-	-	-	-	-	-	-	0.53	-	-	-	-	-
15	fish herbivores	-	-	-	-	-	-	-	-	-	-	-	-	0.09	-	-	-	-	-
16	microphytes	-	-	0.25	0.05	0.05	-	-	0.25	-	-	-	-	-	-	-	-	-	-
17	macroalgae canopy	-	0.10	0.35	0.20	0.60	-	-	0.25	-	-	-	-	-	-	0.24	-	-	-
18	macroalgae foliose	-	-	0.20	0.35	0.15	-	-	-	-	-	-	-	-	-	0.67	-	-	-
19	macroalgae crustose	-	0.20	0.20	0.35	0.15	-	-	-	-	-	-	-	-	-	0.09	-	-	-
20	meso/macrozooplankton	-	-	-	-	-	-	-	-	-	-	0.17	-	-	0.88	-	0.20	-	-
21	microzooplankton	-	-	-	-	-	-	-	-	0.30	0.30	-	-	-	-	-	0.70	0.10	-
22	phytoplankton	-	-	-	-	-	-	-	0.25	0.40	0.40	-	-	-	-	-	0.10	0.65	-
23	bacteria	-	-	-	-	-	-	1.00	0.25	0.30	0.30	-	-	-	0.07	-	-	0.25	0.18
24	detritus	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	0.82

Table 4. 3. Diet Matrix for historic Taputeranga MR ecosystem model. Diets are expressed as a proportion of total diet.

	Prey \ predator	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	20	21	23
1	birds	-	-	-	-	-	$1*10^{-6}$	-	-	-	-	-	-	-	-	-	-	-	-
2	lobster	-	-	-	-	-	$4*10^{-4}$	-	-	-	-	-	-	-	-	-	-	-	-
3	mob inverts herb	0.20	0.13	-	-	-	0.15	-	-	-	-	-	0.18	-	-	-	-	-	-
4	paua	0.03	0.01	-	-	-	0.04	-	-	-	-	-	0.03	-	-	-	-	-	-
5	kina	0.01	-	-	-	-	0.01	-	-	-	-	-	0.01	-	-	-	-	-	-
6	mob invert carn	0.29	0.04	-	-	-	0.14	-	-	-	-	-	0.20	-	-	-	-	-	-
7	sea cucumber	-	-	-	-	-	0.06	-	-	-	-	-	-	-	-	-	-	-	-
8	phytal/infaunal inverts	0.31	0.09	-	-	-	0.10	-	-	-	-	0.58	0.12	-	0.04	-	-	-	-
9	sponges	-	-	-	-	-	0.07	-	-	-	-	-	0.06	-	-	-	-	-	-
10	sessile inverts	-	-	-	0.05	0.05	0.43	-	-	-	-	0.24	0.41	-	-	-	-	-	-
11	fish cryptic	0.16	-	-	-	-	-	-	-	-	-	-	-	0.09	-	-	-	-	-
12	fish inverts	-	-	-	-	-	-	-	-	-	-	-	-	0.21	-	-	-	-	-
13	fish piscivores	-	-	-	-	-	-	-	-	-	-	-	-	0.09	-	-	-	-	-
14	fish planktivores	-	-	-	-	-	-	-	-	-	-	-	-	0.53	-	-	-	-	-
15	fish herbivores	-	-	-	-	-	-	-	-	-	-	-	-	0.09	-	-	-	-	-
16	microphytes	-	-	0.25	0.05	0.05	-	-	0.25	-	-	-	-	-	-	-	-	-	-
17	macroalgae canopy	-	0.24	0.35	0.20	0.60	-	-	0.25	-	-	-	-	-	-	0.24	-	-	-
18	macroalgae foliose	-	-	0.20	0.35	0.15	-	-	-	-	-	-	-	-	-	0.67	-	-	-
19	macroalgae crustose	-	0.49	0.20	0.35	0.15	-	-	-	-	-	-	-	-	-	0.09	-	-	-
20	meso/macrozooplankton	-	-	-	-	-	-	-	-	-	-	0.17	-	-	0.82	-	0.20	-	-
21	microzooplankton	-	-	-	-	-	-	-	-	0.30	0.30	-	-	-	-	-	0.70	0.10	-
22	phytoplankton	-	-	-	-	-	-	-	0.25	0.40	0.40	-	-	-	-	-	0.10	0.65	-
23	bacteria	-	-	-	-	-	-	1.00	0.25	0.30	0.30	-	-	-	0.14	-	-	0.25	0.18
24	detritus	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	0.82

4.3.5 Model Area

I used a backscatter map produced by NIWA using side-scan sonar (Wright et al. 2006) for the Taputeranga MR to delimit the model area. The GIS version of this map makes it possible to determine area of physical bottom type by depth range. In order to estimate bottom type from the backscatter plot, I used ArcGIS software to reclassify the physical parameter 'slope' into 'subtidal reef' and 'subtidal soft and mobile substrates'. In order to ensure that my interpretation of the parameter 'slope' was a valid representation of reality, I ground-truthed the reclassified map with the 'Wellington South Coast Substrates Map' (NZOI 1993). For intertidal regions, I used a GIS version of the 'Wellington South Coast Substrates Map' (NZOI 1993) to determine areas of 'intertidal reef' and 'intertidal soft and mobile substrates'. Because this map does not cover the entire model area, I extrapolated the proportion of 'intertidal reef' to 'intertidal soft and mobile substrates' from the region surveyed to the area of the unsurveyed region. I sub-divided the model area into six regions in order to input region-specific information about the biomass of different species and area of habitat. For each of the six regions, I had information about area of each of the four substrate types. For the subtidal regions, I had information about the amount of area of each substrate type ('subtidal reef', 'subtidal soft and mobile substrates') at 1 m depth intervals. This allowed estimation of biomass for each species for each region in the model area based on substrate type and depth range inhabited.

4.3.6 Model Balancing

To achieve mass-balance of the Ecopath model using initial parameter estimates it was necessary to adjust biomass, production, consumption and diet parameters to produce an ecotrophic efficiency between 0 and 1 for each trophic group. For consuming trophic groups, this ensures that there is enough biomass of their prey items to support the biomass of the trophic group itself. Parameters were adjusted within the range of the confidence intervals according to Pinkerton et al. (2008) for the trophic groups: birds, lobster, mobile invertebrates herbivores, kina, mobile invertebrates carnivores, phytal/infaunal invertebrates, sponges and bacteria. This process gave insight into which trophic groups were consuming a large proportion of prey available to them.

4.3.7 Parameterisation of Fisheries

The most important commercial fishery operating within the model area is the lobster fishery. I used landings data from the CRA4 management area, scaled for the size of the model area to determine annual lobster biomass harvested (Ministry of Fisheries 2009b). The most recent estimate of recreational take of lobster is approximately 10% of commercial landings (Ministry of Fisheries 2009b). This value was used to parameterise the recreational lobster fishery. Important finfish species harvested in the model area are blue cod (*Parapercis colias*) and butterfish (*Odax pullus*). Blue cod is harvested commercially throughout New Zealand, however not within the model boundary (Francis 2008). In terms of biomass, blue cod is the most important species landed by recreational fisheries in New Zealand (Ministry of Fisheries 2009). For recreational catch of this species, I have used recent estimates of recreational harvest from the BCO2 management area scaled to the model area (Ministry of Fisheries 2009). Recreational harvest of butterfish is estimated at approximately 10% of commercial harvest for the management area BUT2 (Ministry of Fisheries 2009). Thus to estimate recreation catch of butterfish from the study area I have applied the percentage to landings from this management area scaled to model area. Paua are harvested recreationally, but not commercially, within the model area and I have used landings estimates for the PAU2 management area scaled to the model area (Ministry of Fisheries 2009).

4.3.8 Reconstruction of Historic Ecosystem

While it is difficult to determine the exact virgin biomass (B_0) of lobster for the model region, there is evidence that lobster biomass in the 1940's was considerably higher than observed today. Breen and Kim (2006) report that vulnerable biomass was approximately four times greater in the 1940's in comparison to the present for the CRA4 management area where the model area is located. I have used this estimate for the historic ecosystem model. In order to reflect the higher biomasses of other species that have also been exploited over the last 60 years, I also increased the biomasses of trophic groups that contain targeted species. In the absence of quantitative information about historic abundances of other exploited species, I doubled biomasses of the trophic groups: kina, mobile invertebrate herbivores, paua and piscivorous fishes. I have increased the biomass of the fish trophic groups; invertebrate feeders, herbivores and planktivores by 1.5 times to reflect the smaller proportion of targeted species that make up these groups. These estimates are based on Ministry of Fisheries virgin biomasses (2009) and could be improved if additional data was available for all exploited species, however for many species this type of data is largely unknown. The 'historic' model refers to the period in the 1940's before large commercial removals of marine species occurred. While there is anecdotal evidence that hapuku (*Polyprion oxygeneios*) inhabited nearshore waters of the Cook Strait prior to intensive exploitation which now restricts is to deeper depths, I could not find any records to give an indication of which areas it was found and at what abundance.

As a consequence of increasing lobster biomass, there is also a corresponding increase in lobster predation on prey items. In order to supply prey biomass for the increased lobster biomass, the diet composition had to be changed to reflect a probable increase in herbivory. This was based on an observation at the Te Tapuwae o Rongokako MR, located near Gisborne, New Zealand following increased lobster density. Lundquist and Pinkerton (2008) reported higher density of lobster inside the MR and a diet that consisted of a greater proportion of macroalgal species than did lobster in neighbouring unprotected areas. I also adjusted the diets of piscivorous fish, planktivorous fish and invertebrate feeding fish in order to provide enough prey biomass for these trophic groups.

4.3.9 Prediction of Future Ecosystem

In order to evaluate the likelihood of the Taputeranga MR ecosystem returning to its historic state, I ran a scenario from its present day state into the future for 40 years using EwE software. The 'future' ecosystem state refers to the results of model predictions for the year 2050 for only the MR portion of the model area. This scenario assumes that there is no fishery harvest from within the model area and does not account for any illegal fishing or poaching. This scenario also does not account for any density-dependent related movement out of the MR that may occur. The purpose of this scenario was to determine if stopping the exploitation of marine resources could return the ecosystem to its historic ecosystem state, which includes higher biomasses of targeted species. Diet proportions were allowed to vary during this scenario in order to allow for changing biomasses of different trophic groups.

4.4 Results

4.4.1 Model Area

The model area is 5428 ha in size, of which the Taputeranga MR comprises 15.7% (854.79 ha; Figure 4.1). 'Subtidal reef' accounts for 580 ha (10%) of model area while 'subtidal soft and mobile substrates' cover 4289 ha (79%). 'Intertidal reef' accounts for 308.5 ha (6 %) of the model area, while 'intertidal soft and mobile substrates' account for 265 ha (5%). Maximum depth within the model area is approximately 100 m, whilst average depth is approximately 25 m. 'Subtidal reef' mostly occurs between 0 to 25 m with a few smaller areas found at deeper depths.

4.4.2 Ecosystem Structure

The ecosystem model for the Taputeranga MR is described by 24 trophic groups linked by 77 predator/prey interactions and resulted in approximately five trophic levels (Figure 4.2; Table 4.4). The majority of biomass within the ecosystem was allocated among primary producers (Figure 4.3). Macroalgal trophic groups accounted for 77% of the biomass in the ecosystem, being made up by 51% canopy, 25% foliose and 2% crustose. Microphytes accounted for 10% of ecosystem biomass. The invertebrate trophic groups accounted for 9% of ecosystem biomass, made up of 2.1% sessile, 2.1% sponges, 1.3% mobile herbivores, 0.8% mobile carnivores, 0.7% phytal/infaunal, 0.6% lobster, 0.3% paua, and 0.1% kina. Detritus made up 1.3% of ecosystem biomass. Plankton made up 1% of ecosystem biomass, being composed of 0.7% phytoplankton, 0.2% meso/macrozooplankton and 0.1% microzooplankton. Bacteria accounted for 0.8% of ecosystem biomass. Fish trophic groups accounted for 0.5%, made up of 0.2% herbivores fishes, 0.1% planktivores, 0.1% invertebrate feeders, 0.1% cryptic reef fishes and less than 0.1% piscivores. Birds accounted for less than

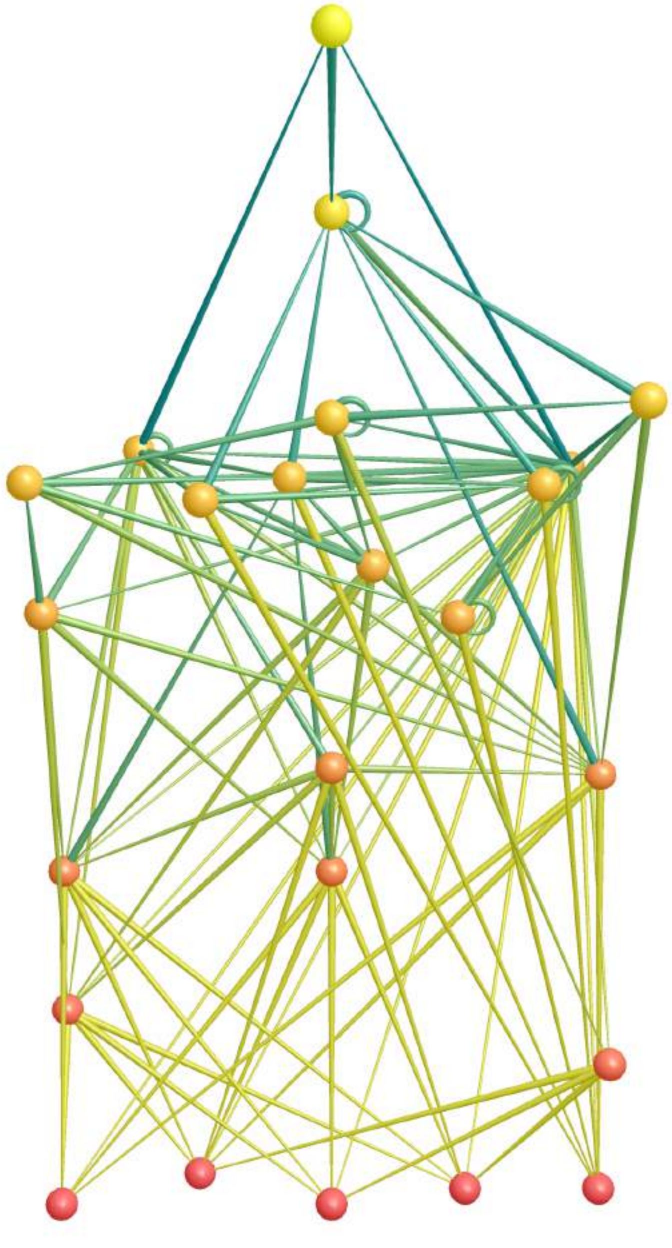


Figure 4. 2. Visualisation of the Taputeranga Marine Reserve food web. Image produced with FoodWeb3D, written by R.J. Williams and provided by the Pacific Ecoinformatics and Computational Ecology Lab (www.foodwebs.org, Yoon et al. 2004). Different colour dots represent functional groups from different trophic levels with red = primary producers, orange = consumers and yellow = top predators. Lines represent feeding links with larger node at predator and smaller node at prey.

Table 4. 4. Outputs by functional group of the ecosystem models of the Taputeranga MR for historic, present and future periods. TL = trophic level; EE = ecotrophic efficiency; M = predation mortality (yr^{-1}).

	Functional group	Past			Present			Future			
		TL	EE	M	TL	EE	M	*F*	*B*	TL	M
1	birds	3.86	0.17	0.02	3.86	0.17	0.02	0.00	0.00	3.92	0.11
2	lobster	2.36	0.002	0.003	3.06	0.98	0.00	0.95	6.36	2.89	0.03
3	mob inverts herb	2.00	0.89	1.16	2.00	1.00	1.30	0.00	0.51	2.00	1.40
4	paua	2.09	0.35	0.52	2.09	0.93	0.76	0.44	0.42	2.12	1.29
5	kina	2.09	0.51	0.56	2.09	0.88	0.97	0.00	0.04	2.11	1.18
6	mob invert carn	3.75	1.00	1.76	3.75	1.00	1.76	0.00	0.50	3.76	1.86
7	sea cucumber	3.22	0.97	0.58	3.22	0.97	0.58	0.00	0.47	3.22	0.57
8	phytal/infaunal inverts	2.30	1.00	3.67	2.30	0.92	3.39	0.00	0.33	2.30	3.92
9	sponges	2.80	0.89	0.18	2.80	0.84	0.17	0.00	1.96	2.78	0.19
10	sessile inverts	2.80	1.00	1.50	2.80	0.88	1.31	0.00	1.81	2.78	1.46
11	fish cryptic	3.64	0.09	0.22	3.64	0.06	0.15	0.00	0.03	3.77	2.40
12	fish inverts	3.77	0.30	0.12	3.89	0.22	0.09	0.00	0.06	3.88	0.42
13	fish piscivores	4.98	0.54	0.23	5.05	0.94	0.23	0.36	0.05	3.84	0.42
14	fish planktivores	4.29	0.37	0.19	4.36	0.28	0.14	0.00	0.12	4.39	0.52
15	fish herbivores	2.00	0.05	0.02	2.00	0.94	0.01	0.82	2.02	2.00	0.09
16	microphytes	1.00	0.04	0.76	1.00	0.02	0.49	0.00	7.77	1.00	20.83
17	macroalgae canopy	1.00	0.12	0.34	1.00	0.06	0.16	0.00	38.10	1.00	2.85
18	macroalgae foliose	1.00	0.03	0.44	1.00	0.02	0.24	0.00	17.97	1.00	13.08
19	macroalgae crustose	1.00	0.34	8.68	1.00	0.11	2.68	0.00	1.30	1.00	25.98
20	meso/macrozooplankton	3.52	1.00	17.67	3.52	0.89	15.75	0.00	0.17	3.52	17.51
21	microzooplankton	2.45	0.97	213.09	2.45	0.97	213.09	0.00	0.06	2.44	227.66
22	phytoplankton	1.00	0.20	65.74	1.00	0.20	65.74	0.00	0.49	1.00	322.41
23	bacteria	2.22	0.98	98.40	2.22	0.98	98.19	0.00	0.60	2.22	99.93
24	detritus	1.00	0.28		1.00	0.28			1.00	1.00	0

0.001% of ecosystem biomass. In the historic ecosystem model, biomass is distributed slightly differently, with 75% accounted for by macroalgae, 11% by invertebrates, 10% by microphytes, 1.3% by detritus, 1% by fishes, 0.9% by plankton, 0.8% by bacteria, and less than 0.001% by birds.

The historic ecosystem model resulted in a 4% greater average biomass per unit area in comparison to the present ecosystem (77.3 g $C.m^{-2}$ vs. 74.5 g $C.m^{-2}$ respectively). The historic ecosystem model differed from the present ecosystem model in higher biomasses for the trophic groups: lobster, mobile invertebrate herbivores, paua, kina, fish – invertebrate feeders, fish – piscivores, fish – planktivores and fish – herbivores (Table 4.1). The diet matrix for the historical and present ecosystem models differed as a result of having greater biomass of some trophic groups in the historical model, which require a greater quantity of prey biomass not supplied by the present diet matrix (Tables 4.2 & 4.3).

4.4.3 Ecosystem Functioning

For the historical and present ecosystem models, the top predators were fish piscivores, with trophic levels of 4.98 and 5.05 respectively (Table 4.4). The future ecosystem scenario predicts that fish planktivores are the top predator with a trophic level of 4.39. For the historical and present ecosystem models, fish planktivores, fish invertebrate feeders and mobile invertebrate carnivores make up the next highest trophic level groups. For the future ecosystem scenario, birds, fish invertebrate feeders and fish piscivores are the groups with the next highest trophic levels. The trophic level of lobster increased from 2.36 in the past ecosystem model to 3.06 for the present ecosystem model as a result of decreased herbivory following the fourfold decrease in lobster biomass (Table 4.4). The future ecosystem model predicts that lobster trophic level decreases to 2.89 as a result of increased herbivory (Table 4.4).

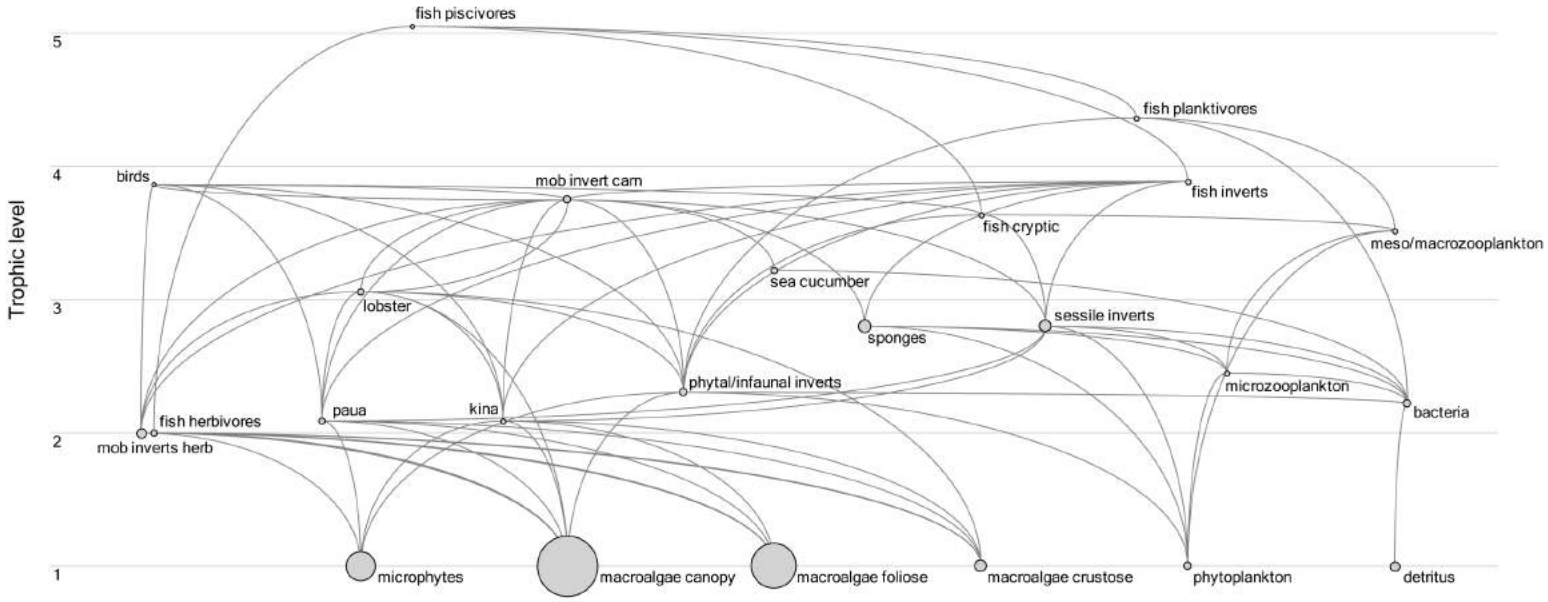


Figure 4. 3. Ecosystem model for the Taputeranga Marine Reserve, New Zealand. Circle size of each trophic group is proportional to amount of biomass. Direction of energy flow is represented by position of line with relation to circle: flows positioned on the top of a trophic group indicate biomass outgoing, while flows positioned on the side indicate entering biomass.

The keystoneness plot indicates that the four most keystone trophic groups for the historic ecosystem model for the Taputeranga MR are invertebrates (Figure 4.4A). Mobile invertebrate carnivores are the most keystone trophic group, followed by lobster, mobile invertebrate herbivores and sessile invertebrates (Figure 4.4A). These groups are followed by piscivorous fishes, macroalgae crustose, phytoplankton, phytal/infaunal invertebrates, microzooplankton, meso/macrozooplankton and birds which round out the top ten. The present ecosystem model for the Taputeranga MR indicates that mobile invertebrate carnivores are still the most keystone trophic group, however lobster have moved from second to eleventh position (Figure 4.4B). The second most keystone trophic group in the present ecosystem model is mobile invertebrate herbivores (ranked third in historic ecosystem), followed by phytoplankton (ranked seventh in the historic model; Figure 4.4).

The mixed-trophic impact analysis displays direct and indirect impacts of very small increases in biomass of groups (impacting groups) on the biomasses of other groups (impacted groups) (Figure 4.5). These impacts are relative, but comparable between groups. For the historic ecosystem model of the Taputeranga MR, mobile invertebrate carnivores were the most keystone trophic group, negatively impacting prey species (excepting mobile inverts herbivores and phytal/infaunal inverts). Lobster were the second most keystone trophic group, resulting in negative impacts for their prey (except macroalgae canopy) and positive impacts on the prey of their prey, indicating a trophic cascade. Mobile invertebrate herbivores were the third most keystone species and negatively impacted their prey species but also competitors who consume the same prey items. Mobile invertebrate herbivores positively impacted three of its predators; birds, mobile invertebrate carnivores and fish – invertebrate feeders.

For the present ecosystem model of the Taputeranga MR, the most keystone trophic group - mobile invertebrate carnivores - has a negative effect on all of their prey species with the exception of lobster. Positive impacts exerted by mobile invertebrate carnivores were observed for the prey of their prey and were of smaller magnitude in

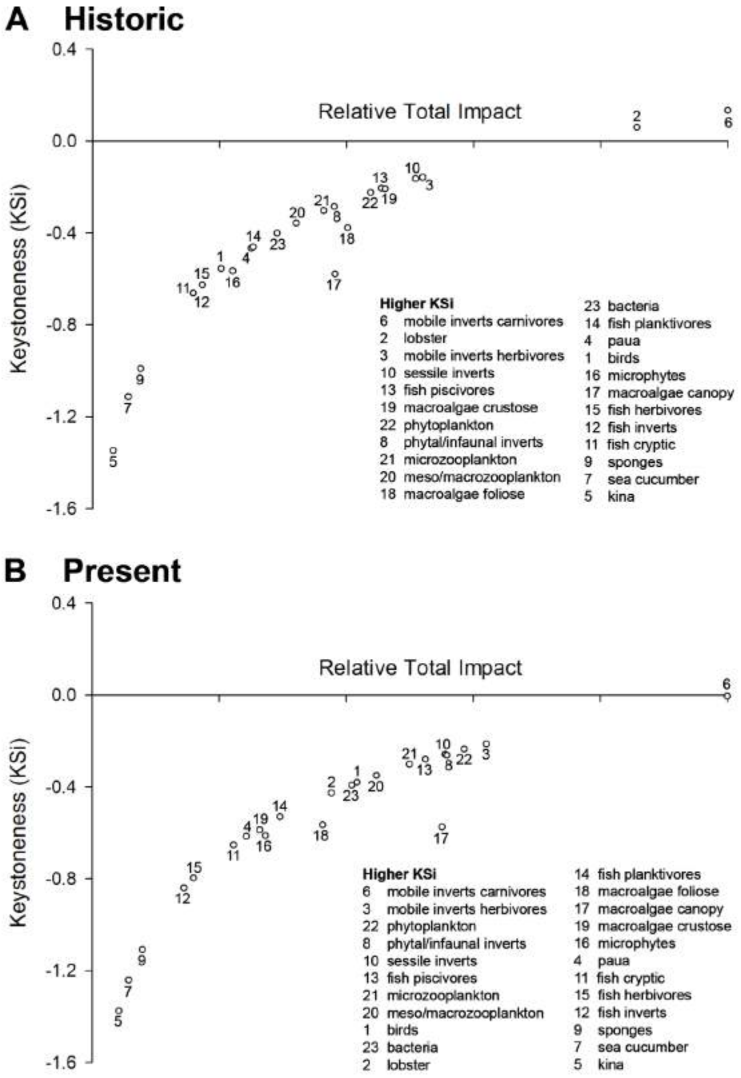


Figure 4. 4. Keystoneness (KS$_i$) and relative total impact (ε_i) of each functional group. Keystone groups are those with higher ε_i and higher KS$_i$ (value close to or greater than zero).

comparison to negative impacts (Figure 4.5). Mobile invertebrate herbivores were the second most keystone group and displayed negative effects on both their prey and also on other grazing trophic groups that compete for the same resources. Phytoplankton was the third most keystone trophic group, however displayed mostly positive effects for impacted groups. This was propagated throughout all trophic levels presumably through increased production flowing through the entire ecosystem. Although lobster are the second most keystone group for the historic ecosystem model, they play a much smaller role in the present ecosystem, most likely explained by their reduced biomass. In the historic ecosystem, macroalgae crustose, phytoplankton and macroalgae foliose are the most important producers, while in the present ecosystem model, phytoplankton is the most important producer group.

The Lindeman spine indicates equal or higher biomass for all of the trophic levels in the historic ecosystem model in comparison to present, with a large difference in the second trophic level (secondary producers; Figure 4.6). The Lindeman spine shows that transfer efficiencies are lower on average for the historic ecosystem model (24.9% for historic and 25.4% for present; Figure 4.6). Ascendancy was lower for the historic ecosystem model at 36.1% in comparison to 36.7% at present.

4.4.4 Impacts of Fisheries and MR Protection

Results from the present ecosystem model indicate that commercial fisheries operating in the model area for lobster and butterfish have the greatest impacts on the ecosystem (Figure 4.5B). Recreationally, the paua and blue cod fisheries have the greatest impacts on the ecosystem (Figure 4.5B). Of the species harvested, lobster requires the most primary production at 1.79%, followed by fish piscivores (0.48%), fish herbivores (0.17%) and paua (0.15%). The majority of biomass taken by catch is from

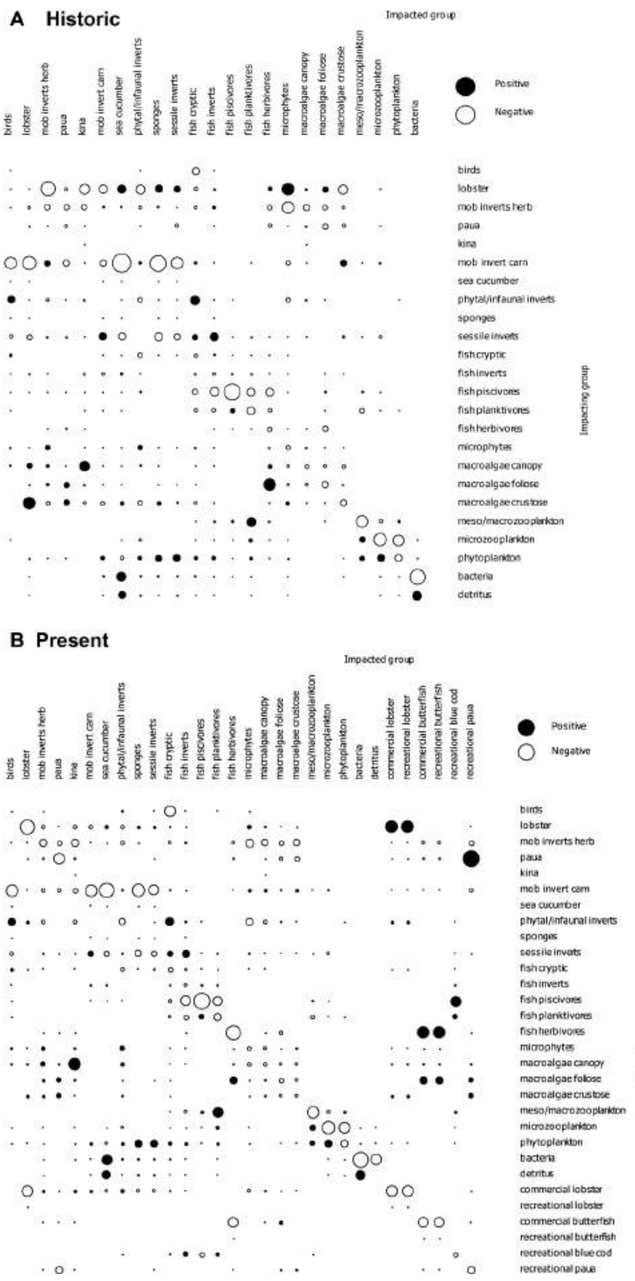
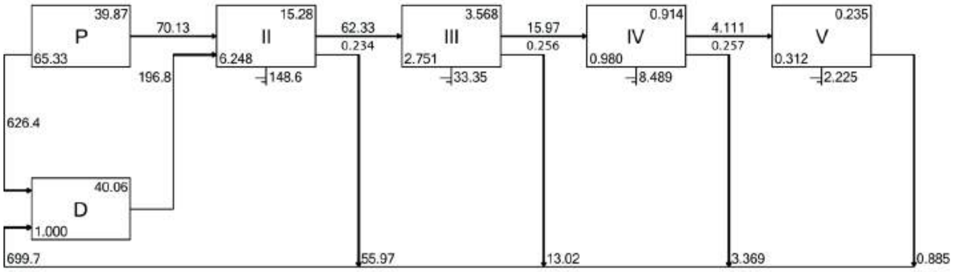
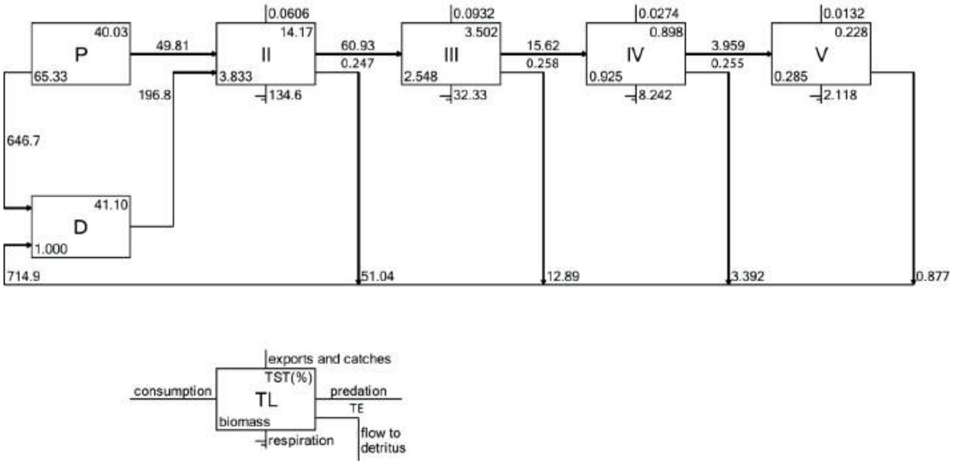


Figure 4. 5. Mixed trophic impacts showing the combined direct and indirect trophic impacts that an infinitesimal increase of any of the trophic groups in rows (impacting group) is predicted to have on the groups in columns (impacted group). Size of the circle is proportional to the degree of change.



A Historic



B Present

Figure 4. 6. Lindeman spine for historic and present ecosystem models showing flows between trophic levels and export of biomass due to catch at each trophic level.

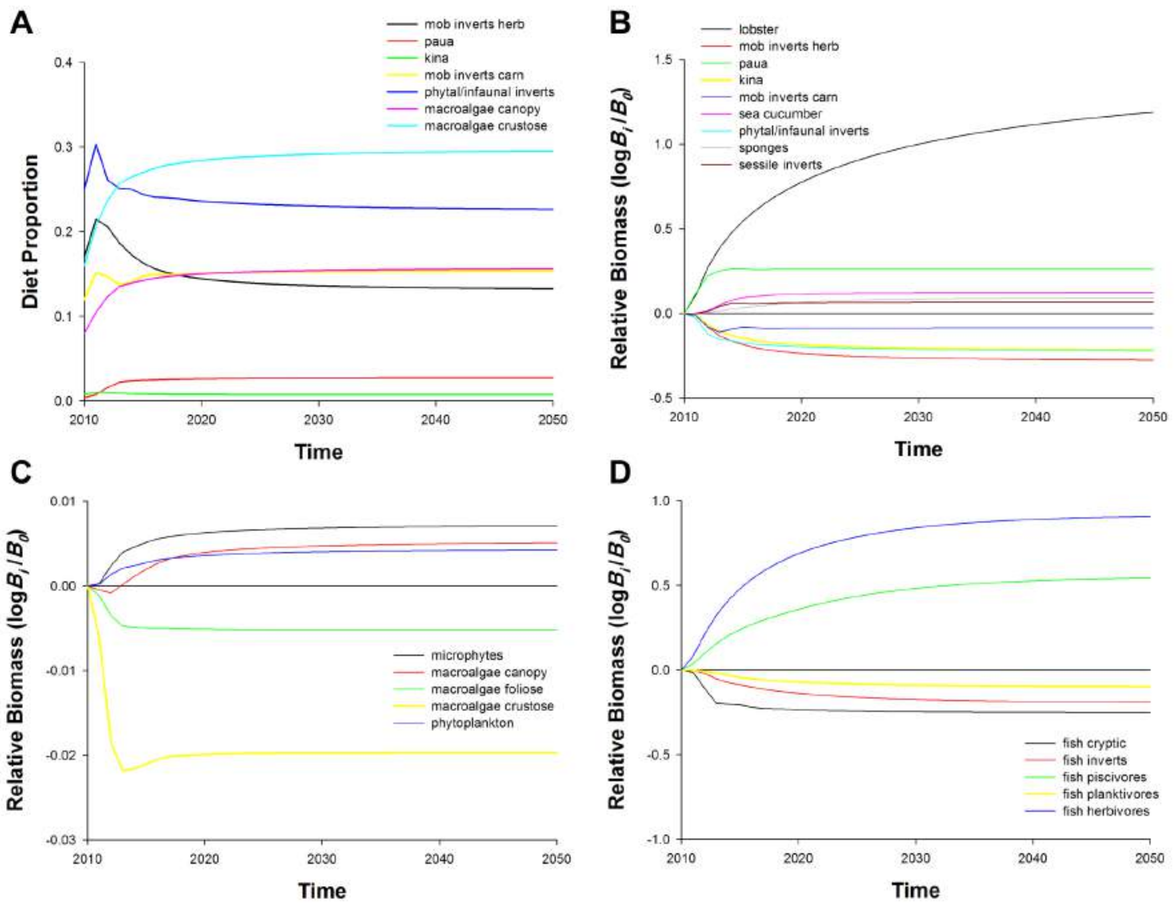


Figure 4. 7. Prediction of lobster diet following MR protection in future scenario (A). Predictions of change in biomass following MR protection for invertebrates (B), primary producers (C) and fishes (D) in future scenario.

secondary producers of trophic level II, followed by trophic levels III, IV, and V (Figure 4.6).

Results from the future scenario for the Taputeranga MR indicate that biomasses of targeted trophic groups are predicted to increase in the absence of fishing (Figure 4.7B, C, D). As a result of increasing lobster biomass, lobster diet changes to include a higher proportion of canopy and crustose macroalgae (Figure 4.7A). This result is similar to diet input for the historic model (Table 1). Overall, total ecosystem biomass is predicted to be 82.1 g C m^{-2} for the future model, which is 10% greater than present and 6% greater than historically occurred. Along with fished trophic groups, other groups that may increase in biomass are sea cucumber, sponges, sessile invertebrates, microphytes, macroalgae canopy and phytoplankton (Figure 4.7B, C, D). Trophic groups that are predicted to decline in biomass in the future scenario are: mobile invertebrates - carnivores, kina, phytal/infaunal inverts, mobile invertebrates - herbivores, macroalgae - foliose, macroalgae - crustose, fish - planktivores, fish - invertebrate feeders and fish - cryptic (Figure 4.7B, C, D).

4.5 Discussion

4.5.1 Modeling Considerations

The purpose of the future ecosystem modeling scenario was to determine if the relative ecosystem structure at present would return to a state similar to that observed in the past following the exclusion of fishery exploitation. The future ecosystem model does not take into account any spillover that may occur from within the Taputeranga MR across the marine reserve boundary to neighbouring fished locations, which is probable given observations of lobster movement from other MRs in New Zealand (Kelly 2001; Kelly and MacDiarmid 2003) and the configuration of the Taputeranga MR boundaries that cross reef habitat patches (Freeman et al. 2009). For this reason, the predicted magnitude of change may be exaggerated, however it elucidates how the ecosystem structure would change in the absence of exploitation. There are many factors that need to be considered when making predictions about future ecosystem states such as: interannual seasonal variability, the El Nino Southern Oscillation (ENSO), climate change, food availability, habitat quality and quantity, reproductive success (larval supply, larval attachment, recruitment), predation pressure, and environmental controls (currents, temperatures). Evidence from other MRs in New Zealand has shown that lobster size and abundance both increase after MR implementation (Pande et al. 2008) and I expect that this will also be the case for the Taputeranga MR, given that the region has supported a productive commercial and recreational lobster fishery for approximately 70 years (Booth 2008).

4.5.2 New Zealand Coastal Ecosystems

The MR induced trophic cascade observed at the Cape Rodney – Okakari Point MR (Leigh or Goat Island) followed recovery of lobster (*Jasus edwardsiii*) and snapper (*Pagrus auratus*; Shears and Babcock 2003). In contrast, the colder waters of the Cook Strait subject to high wave, wind and current energy host a different community of marine species (Shears and Babcock 2007; Pande and Gardner 2008). As a result, trophic control also differs. While commonly sighted in the subtidal waters of the model area, kina (urchins; *Evechinus chloroticus*) do not form large aggregations resulting in kelp barrens (*personal observation*; Pande 2002; Pande and Gardner 2008) that have been observed farther north in New Zealand (Shears and Babcock 2003). In the fished (present) state of the Taputeranga MR, mobile invertebrate carnivores exert the greatest trophic control on the ecosystem. In the historic model, trophic control of the ecosystem is more evenly shared by lobster and mobile invertebrate carnivores.

Compared to the Te Tapuwae o Rongokako MR, the Taputeranga MR also has a high biomass of macroalgae, but has a greater biomasses of most invertebrate trophic groups including threefold more lobster and tenfold more mobile invertebrate carnivores (Pinkerton et al. 2008). Conversely, Te Tapuwae o Rongokako has higher biomasses of three fish groups: invertebrate feeders, piscivores and planktivores. It should be noted that the composition of habitat types for the model area at Te Tapuwae o Rongokako differ, and should be taken into account when making comparisons.

4.5.3 Past, Present and Future Ecosystem States

In the historic and future ecosystem states, lobster biomass is greater due to decreased exploitation, which results in this species having a strong organising role in the ecosystem. The extent to which lobster are able to change their diet and become more herbivorous is largely unknown. Stable isotope analysis of lobster tissue from the Te Tapuwae o Rongokako MR undertaken by Lundquist and Pinkerton (2007) indicated that lobster inside the MR had a higher proportion of macroalgae in their diet than those outside the MR. It was also observed that lobster inside the marine reserve displayed different feeding behaviour than lobster at neighboring unprotected locations by foraging on the intertidal platform at night for a range of macroalgal species (Lundquist, *personal communication*). The Te Tapuwae o Ronokako MR has been in place for 11 years and it has been documented that there is a higher abundance of lobster inside the MR in comparison to neighboring areas (Freeman et al. 2009). It appears that an increased biomass of lobster expected to occur in the Taputeranga MR will result in a dietary change by lobster. The future simulation included increased herbivory on both crustose and canopy macroalgae as this was required to support the larger lobster biomass in the historic biomass.

Commercial and recreational fisheries operating within the model area require a total of 2.59% of primary production from the ecosystem. This figure is less than the global average of 8% (Pauly and Christensen 1995), which is also reported for New Zealand (Knight and Jiang 2009). Predictions from the future scenario indicate that the ecosystem can support increased biomasses of previously targeted species following exclusion of fishing pressure. The lobster stock biomass is being maintained at a level estimated to be approximately one quarter of that in the 1940's (Breen and Kim 2006), indicating that the fishery may be operating below the biomass that produces maximum sustainable yield (BMSY). The dynamics of the ecosystem have also changed as a result of this exploitation, by which mobile invertebrate carnivores are currently classified as the sole keystone trophic group, while in past and future models this role is shared with lobster.

The degree of ecosystem exploitation and degradation that has taken place throughout the last 70 years for the Taputeranga ecosystem is not as severe as has been documented at locations in Europe subject to 2500 years of exploitation (Coll et al. 2008) and is more in line with historical reconstructions that have been undertaken for areas with less history of human exploitation such as northern British Columbia, Canada (Ainsworth et al. 2008). Intermediate to the high levels of ecosystem degradation observed in Europe and low levels in northern Pacific Canada are other ecosystems of North America subject to substantial exploitation since the 1600's (Jackson et al. 2001; Pandolfi et al. 2003; Lotze and Milewski 2004; Rosenberg et al. 2005; Lotze et al. 2006).

However, biomasses of targeted species and trophic groups at the Taputerana MR have been substantially depleted. The model predicts that biomasses of lobster, paua and some fishes will respond positively to MR protection. Species that are highly mobile may not experience increases in biomass, however the overall biomass of the ecosystem protected is predicted to increase and return to a more historic state. The Taputeranga MR that is now in place is predicted to return the ecosystem to a more historic ecosystem state.

Chapter 5

Applying Fishers' Ecological Knowledge to Construct Past and Future Lobster Stocks in the Juan Fernández Archipelago, Chile

5.1 Abstract

Over-exploited fisheries are a common feature of the modern world and a range of solutions including area closures (marine reserves; MRs), effort reduction, gear changes, ecosystem-based management, incentives and co-management have been suggested as techniques to rebuild over-fished populations. Historic accounts of lobster (*Jasus frontalis*) on the Chilean Juan Fernández Archipelago indicate a high abundance at all depths (intertidal to approximately 165 m), but presently lobster are found almost exclusively in deeper regions of their natural distribution. Fishers' ecological knowledge (FEK) tells a story of serial depletion in lobster abundance at fishing grounds located closest to the fishing port with an associated decline in catch per unit effort (CPUE) throughout recent history. I have re-constructed baselines of lobster biomass throughout human history on the archipelago using historic data, the fishery catch record and FEK to permit examination of the potential effects of MRs, effort reduction and co-management (stewardship of catch) to restore stocks. I employed a bioeconomic model using FEK, fishery catch and effort data, underwater survey information, predicted population growth and response to MR protection (no-take) to explore different management strategies and their trade-offs to restore stocks and improve catches. My findings indicate that increased stewardship of catch coupled with 30% area closure (MR) provides the best option to reconstruct historic baselines. Based on model predictions, continued exploitation under the current management scheme is highly influenced by annual fluctuations and is unsustainable. I propose a community-based co-management program to implement a MR in order to rebuild the lobster population while also providing conservation protection for marine species endemic to the Archipelago.

5.2 Introduction

As a response to reports of declining and unsustainable fisheries worldwide (Coll et al. 2008; Agnew et al. 2009; Mora et al. 2009; Pitcher et al. 2009; Worm et al. 2009) there has been much debate among conservationists, fisheries biologists and fisheries managers (Stokstad 2009) about the best means to balance sustainable exploitation with conservation of biodiversity and ecosystems. Proposed solutions include, but are not limited to, ecosystem-based management, MRs and other forms of fishery closures, incentives, co-management, total allowable catch (TAC) and individual transferable quotas, reductions in fishing fleet capacity and changes in gear regulations (Pitcher 2001; Pauly et al. 2002; Browman et al. 2004). Elsewhere it has been suggested that the tools for effective management of fish stocks are already available and that fishery science is sound, but that recommended harvest limits are rarely implemented as policy because of lobbying by stakeholders (Rosenberg 2007). I examine co-management strategies in Chile where the stakeholder group that most often objects to fisheries regulations, fishers, has taken a major role in the management of their livelihood. In the absence of information about the response of lobster to MRs in Chile, I examine the potential of MRs for fisheries management in the Juan Fernández Archipelago using observations from New Zealand. I then investigate the effects co-management, MRs and traditional fisheries management tools for their effectiveness to rebuild an overexploited Chilean lobster (*Jasus frontalis*) fishery as well as promote conservation values and ecosystem protection.

5.2.1 Marine Reserves and Co-Management

In Chile and New Zealand, MRs are implemented for conservation purposes, but may produce indirect benefits for fisheries because they have been shown to increase the size, abundance and biomass of many fished species, including the New Zealand lobster, *Jasus edwardsii* (Kelly et al. 2000; Davidson et al. 2002; Shears et al. 2006; Pande et al. 2008), to safe-guard against fishery-associated handling disease (Freeman and MacDiarmid 2009), and to increase population-specific egg production rates because larger lobster produce disproportionately more eggs than smaller lobster (Kelly et al. 2000). However, while the area of the MR may benefit from a reduction in fishing pressure, adjacent areas may not. For example, the implementation of a MR often displaces fishing effort, resulting in greater fishing effort per unit area outside the MR (Hilborn 2002; Murawski et al. 2005; Greenstreet et al. 2009; Powers and Abeare 2009). A concern often voiced by fishers is that if the MR does not benefit the region by providing lobster via spillover, then CPUE will be lower in areas adjacent to MRs. However, this need not be the case as CPUE at locations adjacent to a MR and locations further afield may be similar, although the catch made surrounding the MR may be represented by fewer, larger lobster resulting in a similar amount of profit per trap haul (e.g., Kelly et al. 2002). Lobster protected by a MR as small as 400 ha have increased in density, with larger adults making periodic movements across the MR boundary where they “spillover” to the fishery (Kelly et al. 2002), a phenomenon influenced by the position of MR boundaries in relation to rocky reef habitat because lobster are less likely to cross soft sediment habitat (Freeman et al. 2009).

Co-management between fishery managers and fishers has resulted in several benefits in Chile, including (1) it changes the nature of fishing as fishers become stewards of the resource and catches become more predictable (Gelcich et al. 2005; Gelcich et al. 2007); (2) compliance is greater in a community-managed system where local stakeholders have a vested economic interest in the welfare of the resource

(Castilla and Fernandez 1998); (3) it increases the conservation ethic of fishers with greater conservation-oriented attitudes correlated to a longer involvement with co-management (Castilla and Fernandez 1998; Gelcich et al. 2008); (4) it increases biodiversity in co-managed areas (Gelcich et al. 2008) and (5) it may act as a bridge to implement further conservation actions such as MRs. However, in order for a system of co-management to experience high compliance, fishers need be an integral part of the management process which strives to achieve goals set by community (Gelcich et al. 2006; Blaikie 2006; McClanahan et al. 2006; Sutinen and Kuperan 1999; Cinner et al. 2008; Pollnac et al. 2010). A bioeconomic evaluation of co-management needs to include social, economic and biological components; without all of these elements the system becomes oversimplified (Wilen et al. 2002).

In order to evaluate sustainable fishery management options, it is necessary to determine the current level of stock exploitation. Comparison of stock biomass at present time to "virgin" biomass (biomass under an exploitation rate of 0) indicates how exploited a stock has become (Hilborn and Stokes 2010). Stocks that are fished below the biomass that produces maximum sustainable yield (BMSY) are less productive and it maybe desirable from both economic and ecological perspectives to rebuild the stock to a historic, more abundant state. However, it may be difficult to determine if a stock has been fished below BMSY if there is a lack of information about historic stock abundance. This can lead to what has been dubbed "the shifting baseline syndrome" resulting in a distorted view of what is "virgin" biomass (Pauly 1995). In the absence of stock abundance estimates over time, alternate techniques employing historical knowledge from non-scientific sources are needed to place the current state of stock abundance in a larger context (Pitcher 2001; Rosenberg et al. 2005; Ainsworth et al. 2008; Thurstan and Roberts 2010). Historic accounts and fishers' ecological knowledge (FEK) are information sources that can provide insight into changes in stock abundance on inter-generational time scales and prior ecosystem states (Pitcher 2001; Johannes et al. 2000).

5.2.2 The Chilean Juan Fernández Lobster Fishery

My study site, the Chilean Juan Fernández Archipelago, is located in the south Pacific Ocean (33° 37´S, 78° 51’ W), 700 km west of the port city of Valparaíso (Figure 5.1). The volcanic islands that make up the archipelago (Robinson Crusoe, Santa Clara and Alexander Selkirk) display a high degree of endemism in both terrestrial and marine environments (Rozbaczylo and Castilla 1987; Santelices 1992; Pequeño and Lamilla 2000); this applies to the lobster, *Jasus frontalis*, found only on the Juan Fernández Archipelago and the Islas Desventuradas (Figure 5.1). The lobster fishery is the main source of economic revenue for fishers inhabiting the Archipelago. The decline in lobster abundance and the associated change in its natural distribution are documented in historic accounts (Walter 1776; Molina 1808; Albert 1898; Skottsberg 1956). Bahía Cumberland is the main fishing port of the Juan Fernández Archipelago from which

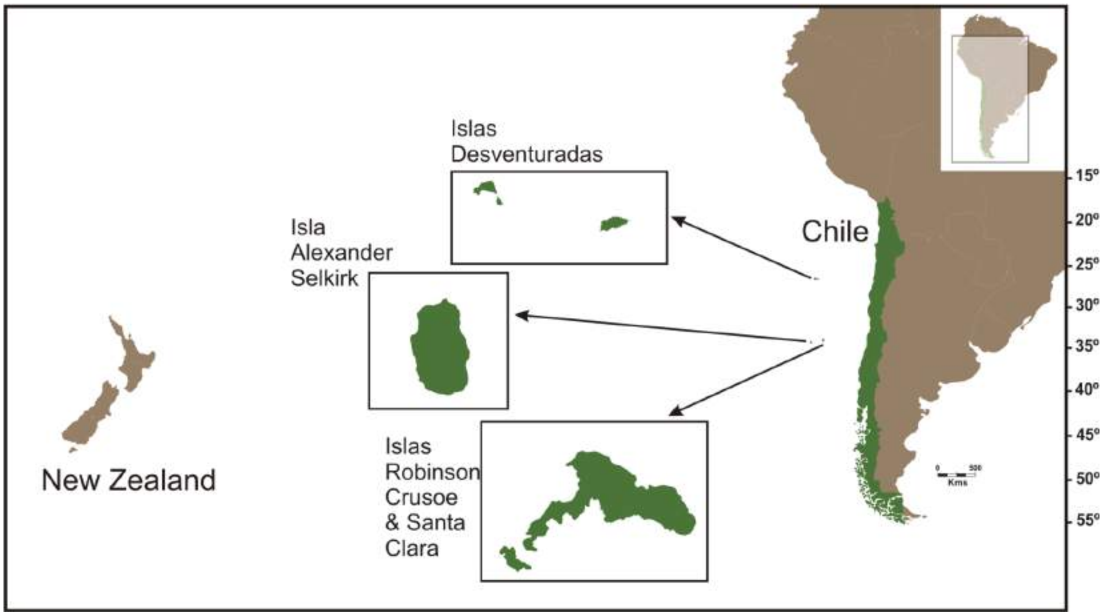


Figure 5. 1. Location of Juan Fernández Archipelago and Islas Desventuradas in relation to Chile and New Zealand. Islas Robinson Crusoe, Santa Clara and Alexander Selkirk collectively make up the Juan Fernández Archipelago.

approximately 180 fishers operate approximately 40 boats to fish at Islas Robinson Crusoe and Santa Clara (Sernapesca 2009). Additionally, seasonal fishing camps are set-up on Alexander Selkirk Island. Wooden boats between 8 to 10 m in length are powered by an outboard motor, and sometimes fitted with a winch and depth finder (Figure 5.2A; Arana and Ziller 1985). Lobster fishing practices using wooden traps (1.35 m by 0.78 m by 0.37 m; Arana and Ziller 1985) have remained relatively traditional (Figure 5.2B).

The current lobster management regulations include a seasonal closure from May 15th until September 30th, a minimum cephalothorax harvest size of 11.5 cm and no capture of egg-carrying females, however there are no catch limits. An informal management system exists whereby location of trap placement is governed by a complex, highly structured system with high compliance (Ernst et al. 2010). Because it is based on the use of traps, the fishery itself is selective, but several finfish species are caught for bait via other methods, resulting in the harvest of approximately 150 kg of fish per day (personal communication with local fishers). The current lobster fishing effort on Islas Robinson Crusoe and Santa Clara is concentrated in the farthest reaches of the archipelago in relation to the population centre and main port, Bahía Cumberland (Ernst et al. 2010). Fishers who camp on Isla Alexander Selkirk (the most isolated region of the Archipelago) report a much higher CPUE (Arana and Vega 2000). At present, STIPA Juan Fernández (the Juan Fernández fishers' syndicate) is a fishers' union concerned with the marketing and management of lobster with a mandate for the conservation and sustainable management of marine biodiversity within the archipelago. The lobster fishery on Juan Fernández is classified by the Chilean government as “artisanal” which gives exclusive fishing rights to registered fishers on the archipelago from land to five nautical miles offshore and prevents new fishers from entering the fishery.

Figure 5. 2. Lobster fishing gear on the Juan Fernández Archipelago. A - Technology consists of a wooden boat, gas-powered winch, depth finder and outboard engine. Photograph by Alejandro Perez-Matus. B - Wooden lobster traps. Photograph by Fabian Ramírez.

5.2.3 Lobster Stocks: Past and Future

My research focuses on Islas Robinson Crusoe and Santa Clara due to limited access to, and lack of availability of information for, Isla Alexander Selkirk. Combining FEK, underwater observations and collection of historic, government and scientific information for Islas Robinson Crusoe and Santa Clara, my aims are three-fold. First, I estimate baselines of lobster biomass over 400+ years of human fishing activity on the archipelago which has led to the current lowest recorded catches in history. Second, I develop a bioeconomic fishery model to describe the dynamics of lobster abundance and the catch record throughout the last century using biological parameters from the closely related lobster species, *Jasus edwardsii*, in the absence of such biological data for *J. frontalis.* Third, I use the bioeconomic model to predict how differing management strategies will impact both lobster abundance and fishery catch over the next 40 years to restore stocks and promote a sustainable fishery. Overall, I describe a 500-year period of lobster exploitation and assess the trade-offs between catch and conservation in the social context of the artisanal fishing community at the Juan Fernández Archipelago. I demonstrate that this technique of reconstructing baselines utilising biological, historic and social information is a powerful tool to understand the relationship between prior and current stock states when considering future management options for conservation and sustainable exploitation of coastal resources.

5.3 Methods

5.3.1 Reconstructing Baselines

I refer to the period of early human exploitation of the marine resources of the Juan Fernández Archipelago before large numbers of lobster were removed from the population as the “virgin” period (1574-1898). As the intertidal zone and shallow subtidal depths were fished of lobster (Walter 1776; Molina 1908; Albert 1898; Skottsberg 1956), effort moved to deeper waters, which we term the “historic” period (1898-1930). The fishery catch record begins in 1930 for all landings of *Jasus frontalis* in Chile (including Isla Alexander Selkirk and the Islas Desventuradas). The proportion of catch from Islas Robinson Crusoe and Santa Clara was estimated to be 65% (± 5% standard deviation) from 1972 – 1983 (Yáñez et al. 1985) and 49% for the 1996 - 1997 season (Arana and Vega 2000). I used these values to model a catch record for Islas Robinson Crusoe and Santa Clara which show landings varying by as much as 46 tonnes between successive years (e.g., 1942 versus 1943). Although highly variable, the average catch remained stable at approximately 60 tonnes per year until 1967. I call 1930-1967 the “fishing” period. From 1967 until 1982, catches declined despite evidence of increasing fishing effort (Yáñez et al. 1985), after which they leveled off and reached a new average catch of approximately 20 tonnes per year. Lobster catch for all of the island groups was declared to be 1 tonne in 2004 when the fishery was closed by the Chilean National Fishery Service (SERNAPESCA) for one season to allow stocks to recover, resulting in the 2005 and 2006 seasons producing the highest catches in 30 years. The last year for which I have catch data is 2008. Based on the low catch and increasing effort I define the years 1967-2008 as the “over-fishing” period. I identify these four designations in the history of lobster fishing on the Juan Fernández Archipelago because each period represents a different state for lobster in terms of biomass. I use information from these separate periods to facilitate the calculation of average baselines and thereby model development for the “fishing” and “over-fishing” periods, to allow me to investigate how alternative management strategies might influence what I call the “future of fishing” period from 2008-2050.

I used the most recently published stock assessment and composition of catch by depth data (Arana and Vega 2000; Yáñez et al. 2000) to estimate the lobster biomass during the “over-fishing” period. In the 1996-97 fishing season, the Robinson Crusoe – Santa Clara stock was estimated to be approximately 70 tonnes (Yáñez et al. 2000) and the reported catch averaged over those two years was 34 tonnes (Sernapesca 2009) indicating that approximately 50% of available stock was harvested (Arana and Vega 2000). I used this value of 50% to calculate lobster biomass for the average catch made during the “over-fishing” period in which most traps were set in water between 112 and 165 m deep (Arana and Vega 2000). For the “fishing” period, I assumed that catch represented a smaller proportion of total stock biomass because CPUE was higher during this period (Yáñez et al. 1985). Based on the historic accounts of biomass and depth of “best catch” (Walter 1776; Molina 1808; Albert 1898; Skottsberg 1956), CPUE (Arana and Vega 2000; Arana and Toro 1985), and area of habitat by depth, I estimated lobster biomass for the “historic” and “virgin” periods.

5.3.2 Quantifying Spatial and Temporal Trends

Conversations with elders of the Juan Fernández Archipelago fishing community (FEK; Johannes et al. 2000) provided information about spatial changes in lobster abundance and fishing effort throughout recent history that I could not find in government or literature sources. These fishers have been fishing on the Juan Fernández Archipelago for as many as 40 years, providing information dating back to 1967 in some cases, corresponding to the transition from the “fishing” to the “over-fishing” period. Anecdotal information about the spatial and temporal distributions of fishing effort, catches and lobster abundances throughout the archipelago were recorded as either “high”, “moderate” or “low”. This information was used to understand how lobster abundance, catch and effort have changed in the archipelago during the last 40 years and was used during development of spatial dynamics for the model described below.

In order to quantify fishers' observations of current lobster abundance distributions at shallow depths (< 30m), I conducted underwater surveys of invertebrate abundance during the months of September and October, 2007 (*manuscript in review*). I selected sites around Isla Robinson Crusoe and Isla Santa Clara to sample separate regions of the archipelago with different subtidal habitats and wave exposure (Figure 5.3A). Wave-exposed sites located on the western side of the archipelago are characterised by vertical walls, big boulders and caves, whereas eastern sites are more gradual in slope, highly eroded and characterised by sand, small boulders and cobble. At each site, a 0.25 m^2 quadrat (50 x 50 cm) was placed at 4 m intervals on both sides of a 20 m transect (10 quadrats per transect) recording the abundance of invertebrate species including lobster. Between six and ten transects were completed at each site (mean = 6.5 ± 2.3 standard deviation; 39 total) based on depth and weather conditions. Transects were conducted perpendicular to the shore to survey a range of depths at each site. Lobster abundance per site is expressed as a percentage of total abundance at all sites, standardised for area surveyed.

5.3.3 Bioeconomic Fishery Model

I have employed a Schaefer biomass dynamic fishery model (Schaefer 1954; equation 1), an economic and behavioural fishery model (Ruth and Lindholm 2002; equation 2) and a biological movement model in a spatial context (Ruth and Lindholm 2002; equation 3).

$$\frac{dB}{dt} = rB(1-\frac{B}{k}) - C$$

(equation 1)

$$\frac{dC}{dt} = \alpha(PqEB - JE)$$

(equation 2)

$$\frac{dB_i}{dt} = m_j\beta_jB_j - m_i\beta_iB_{i-inside}$$

(equation 3)

These models are combined to produce a bioeconomic fishery model (equation 4), described by the terms: B_i = biomass of lobster in region *i*, *t* = time, *r* = intrinsic rate of population growth, k_i = carrying capacity in region *i*, C_i = catch in region *i*, α = rate at which changes in fishing effort take place, *P* = price, q_i = effectiveness of fishing effort (catchability) in region *i*, E_i = fishing effort in region *i*, J_i = travel cost to fishing grounds for region *i*, m_i = movement rate of lobster biomass in region *i*, β_i = uniform distribution variable in region *i*, and ε = annual variability.

$$\frac{dB_i}{dt} = rB_i(1 - \frac{B_i}{k_i}) + (m_j\beta_jB_j - m_i\beta_iB_i) - \alpha(Pq_iE_iB_i - J_iE_i) + \varepsilon_t$$

(equation 4)

I use the term *j* to refer to regions neighboring region *i* in a spatial context. The uniform distribution probability (β) is a randomly generated value between 0 and 1 to simulate the probability of lobster biomass moving to a neighbouring region, given a specific movement rate and lobster density for the region of origin. I chose the Schaefer biomass dynamic model (equation 1) because of the absence of age-structured data for the time scale of the fishery catch record I examined (Hilborn and Walters 1992).

I have incorporated the economic terms price (*P*) and cost (*J*) in the model (equations 2 and 4) to reproduce observations by fishers (FEK) of changes in stock abundance both spatially and temporally. Using these terms in the model allows for areas closest to Bahía Cumberland (the main port) to be fished first given sufficient lobster density. I use the α term (responsiveness to changes in stocks) in order to model fishers' behaviour to changes in stocks. Following a season with a lower catch, fishers fish less in order to maintain stock biomass, a strategy employed in Chilean MEABRs (Gelcich et al. 2005; Castilla and Fernandez 1998; Meltzoff et al. 2002): I refer to this behaviour as "stewardship". The model assumes that fishing technology remains constant during each period (fishing, over-fishing, future of fishing) which is defined as catchability (q) in the model.

By designating separate spatial areas (Figure 5.3B; i and j), I was able to input region-specific lobster biomass (B_i), fishery catch (C_i), travel cost (J_i) and movement rates (m_i), as well as simulate areas designated as MRs (no-take) to predict associated density-dependent spillover effects. Six spatial areas were designated based on lobster abundances and habitat classifications observed during underwater surveys (Figure 5.3A and B), locations of historical and current lobster abundance and fishing efforts (FEK), and accessibility to different regions of the archipelago. I have allowed for the movement of lobster between neighbouring regions in the model according to their density and movement rate (m_i, m_j), multiplied by a uniform distribution probability (β_i, equation 4). This feature of the model allows for "spillover" of lobster from areas of higher density to areas of lower density, a phenomenon that has been documented in New Zealand for *Jasus edwardsii* (Kelly et al. 2002; Freeman et al. 2009), a species which shares many biological characteristics with *Jasus frontalis* (Annala et al. 1980; Arana et al. 1985; Diaz and Arana 1985; Annala and Bycroft 1987; Booth and Ovenden 2000; Dupré 2000). As a consequence, fishing effort changes spatially in response to lobster biomass (Bi), cost of travel (J_i) and fishers' responsiveness to changes in lobster biomass (α; equation 4).

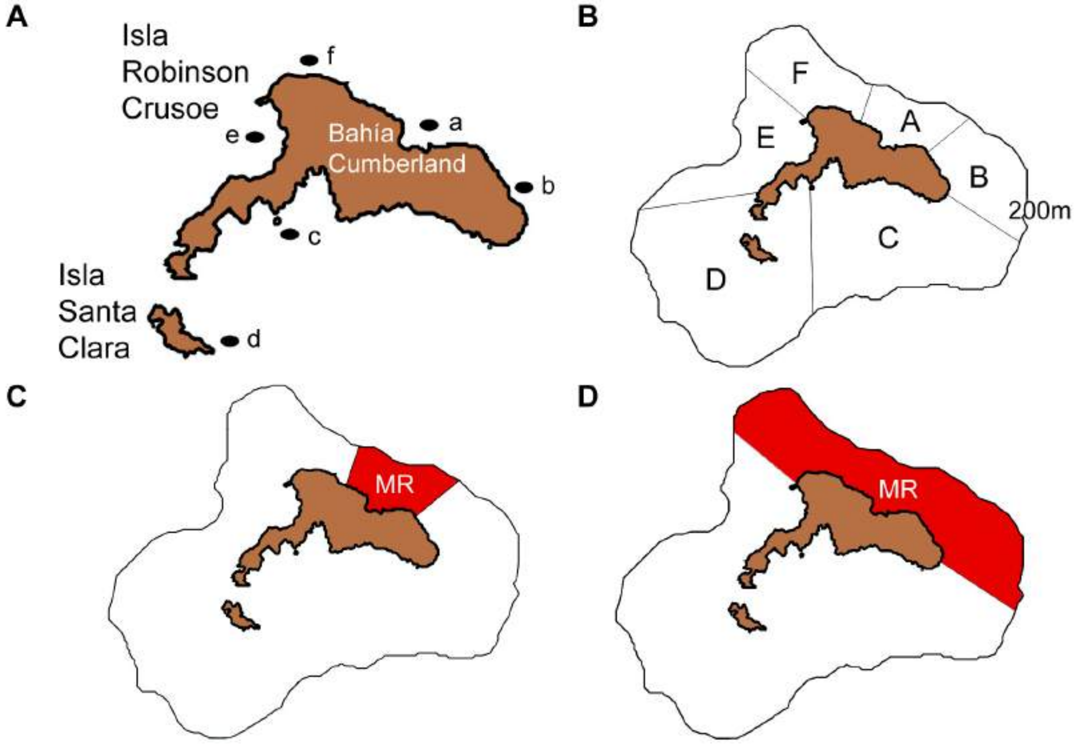


Figure 5. 3. Map showing underwater survey locations, spatial regions and MR locations. A – Map of Islas Robinson Crusoe and Santa Clara showing the main fishing port of Bahía Cumberland and underwater survey locations (a – f; see Table 1 for names). B – Spatial regions used in model scenarios (A – F) with 200m depth contour. C – Location of 10% MR used in model scenarios. D – Location of 30% MR used in model scenarios.

5.3.4 Parameter Estimation

Given the large timescale and the limited amount of data, I used a variety of sources and techniques to estimate parameters (Annala et al. 1980; Arana et al. 1985; Diaz and Arana 1985; Annala and Bycroft 1987; Booth and Ovenden 2000; Dupré 2000; Table 5.1). Intrinsic rate of increase (*r*) was estimated from two sources; first, from a literature value for *Jasus frontalis* (Yáñez et al. 1985), and second, using data for the recovery of 14 *Jasus edwardsii* populations following the reduction of fishing pressure in New Zealand after MR implementation (Pande et al. 2008). Both values were calculated for a highly exploited stock biomass indicating that they should be accurate values of growth for my model and only varied by 0.06% (Hilborn and Walters 1992). Biomass dynamic models are sensitive to intrinsic rate of population growth (*r*) at low biomass (Hilborn and Walters 1992), however I have a high confidence in my value for the model due to the similar values given by empirical evidence for both *Jasus edwardsii* and *Jasus frontalis* (Yáñez et al. 1985; Pande et al. 2008). Carrying capacity (*k*) was estimated using historic accounts of lobster density (Walter 1776; Molina 1808) and extrapolated to area of suitable habitat in each *i* region. Initial lobster biomass was estimated using the catch record, reports of CPUE (Arana and Vega 2000; Yáñez et al. 1985), stock assessments (Yáñez et al. 2000), FEK, and historic accounts. Effort was determined spatially using accounts of FEK and data from the 1972-1983 period (Yáñez et al. 1985) and the 1996-1997 season (Arana and Vega 2000). Movement rate was calculated from tagging and MR spillover studies for *Jasus edwardsii* (Kelly et al. 2002; Kelly 2001; Kelly and MacDiarmid 2003) and adjusted for the area of each of the six spatial areas (smaller area = greater chance of emigration). Price of lobster and travel cost per unit lobster were chosen such that patterns of fishing predicted by the model were similar to those reported by the fishers over time. For each of the "fishing" and "over-fishing" periods, four free parameters (α, ε, *P* and *q*) were estimated by minimising residual sums of squares (RSS) in comparison to observed data (Burnham and Anderson 2002) and were used for the "future of fishing" period.

5.3.5 Model Validation and Prediction

I confronted competing bioeconomic models composed of varying numbers of the four free parameters (α, ε, *P* and *q*) with the observed catch data for the *a priori*

Table 5. 1. Model parameters and initial conditions for the three periods; Fishing, Over-fishing and Future of fishing. State variables and parameters that were spatially resolved are indicated by *.

State variables, Parameters and units	Fishing Period (1930 – 1967)	Over-fishing Period (1967 – 2008)	Future of Fishing Period (2008 – 2050)
Lobster population			
growth rate – r	0.0672	0.0672	0.0672
carrying capacity – k (kg)	400 000	400 000	400 000
initial biomass* – B (kg)	200 000	116 000	61750
movement rate* – m_i	0.04 – 0.125	0.04-0.125	0.04-0.125
uniform distribution variable - β	0 - 1	0 - 1	0 - 1
Fishery catch			
initial effort* – E (number of fishing trips)	805	2414	1207-2414
stewardship - α	0.001	0.0012	0.001- 0.01
catchability – q	0.00001	0.00003	0.00003
price – P	20	20	20
cost – J_i *	4-10	5-10	5-10
annual variation - ε (standard deviation)	coswave (30 000, 4)	sinwave (20 000, 4)	sinwave (20 000, 4)

defined “fishing” and “over-fishing” periods. Model iterations were integrated using the Runge-Kutta 4 method with a time step of 0.125 years and were run for 37 years for the “fishing” period (1930 - 1967), 41 years for the “over-fishing” period (1967 – 2008) and 42 years for the “future of fishing” period (2008 – 2050) using STELLA software (Ruth and Lindholm 2002). I used an Akaike Information Criterion (AIC) approach (Akaike 1973) to assess competing model performance:

$$AIC = N\ln\left(\frac{RSS}{N}\right) + 2K$$

(equation 5)

where *N* is the sample size, *K* is the number of model parameters and *RSS* is the residual sums of squares. Lower *AIC* values indicate a better level of model support (Akaike 1973).

The “future of fishing” (2008-2050) model employed scenarios with regions designated as MRs (closed to the fishery), fishing effort reduction (ER) and stewardship, as well as “business as usual”, indicating no change in management practice. The “future of fishing” model utilised the same optimised parameters (α, ε, P and q) as the “over-fishing” period. Eight different scenarios were run for 42 years with variable amounts of fishing effort and area, with or without stewardship of catch. The 10% MR (10% of fishing grounds closed) is centred in Bahía Cumberland (Figure 5.3C), which was suggested by lobster fishers to be the best location because it is the area most depleted in abundance and most easily enforced and monitored by “the eyes of the village”. The placement of the 30% MR (30% of fishing grounds) is centred in Bahía Cumberland as previous, but extended to the east and west to include El Francés and Sal si puedes (Figure 5.3D). These three regions are the least fished, with the lowest number of traps set throughout the archipelago, that is, 15.6% of all traps in 30% of the area (Arana and Vega 2000).

5.4 Results

5.4.1 Historic Baselines

Visitors to the Juan Fernández Archipelago in the 1700's found that lobster were "… in such abundance near the water's edge (of Isla Robinson Crusoe) that the boat-hooks often struck into them, in putting the boats to and from the shore" (Walter 1776) and were " … found in such quantities that the fishermen have no other trouble than to strew a little meat upon the shore, and when they come to devour this bait, as they do in immense numbers, to turn them on their backs with a stick…" (Molina 1808). Almost one hundred years later, lobster "… were fished at depths of 7 to 14 m …" (Albert 1898, p. 6), while fifty years after this "… the best catch is made in depths from 40 to 80 m ..." (Skottsberg 1956, p. 178). The most recent study during the 1996-1997 season found that the highest number of lobster per trap occurred between depths of 112 to 165 m at Islas Robinson Crusoe and Santa Clara with a CPUE of 10 lobster per trip compared to 32 per trip at Isla Alexander Selkirk, and 174 per trip at Islas Desventuradas (Arana and Vega 2000). Historic lobster abundance estimates in the intertidal and shallow subtidal zones described by Walter (1776) and Molina (1808) are substantially different from those described by Albert (1898) and Skottsberg (1956). FEK and underwater observations show that the majority of lobster abundance is currently concentrated in the farthest reaches of the Archipelago (Table 5.2).

Based on my reconstruction of baselines at Islas Robinson Crusoe and Santa Clara, I have estimated lobster biomasses of 400 tonnes for the "virgin" period and 350 tonnes for the "historic" period (Figure 5.4). Based on stock assessments and reports of CPUE I have estimated average lobster biomasses of 160 tonnes for the "fishing" period and 62 tonnes for the "over-fishing" period (Figure 5.4). My model results for the management scenarios predicted lobster biomasses of 62 tonnes for "business as

Table 5. 2. Fishers' ecological knowledge recorded by region from lobster fishers and results of underwater survey for *Jasus frontalis* (% of total abundance) at sites on the Juan Fernández Archipelago. Refer to Figure 5.3 (panels A & B) for location of sites and regions.

Site or region	Previous abundance	Current abundance	Previous fishing effort	Current fishing effort	% of total abundance
A - Bahía Cumberland	Moderate	Low	High	Low	3.1
B - El Francés	Moderate	Low	Moderate	Low	16.1
C - Los Chamelos	High	Moderate	Low	Moderate	6.3
D - Santa Clara	High	Moderate	Low	High	61.9
E - El Cernícalo	High	Moderate	Low	Moderate	9.4
F - Sal si puedes	Moderate	Low	Moderate	Low	3.1

usual", 140 tonnes for "10% MR", 105 tonnes for "50% ER", 160 tonnes for "10% MR & 50% ER", 200 tonnes for "30% MR", 113 tonnes for "stewardship", 185 tonnes for "stewardship and 10% MR" and 235 tonnes for "stewardship and 30% MR" (Figure 5.4).

5.4.2 Model Selection

The model that was best able to predict the "fishing" period catch data also best described the "over-fishing" period catch data (Table 5.3). Inclusion of the annual variability term improved model fit to fishery catch data. For the "fishing" period, the model was able to predict the annual cycles in lobster catch, but not to the same magnitude of fluctuation as was actually observed (Figure 5.5). For the "over-fishing"

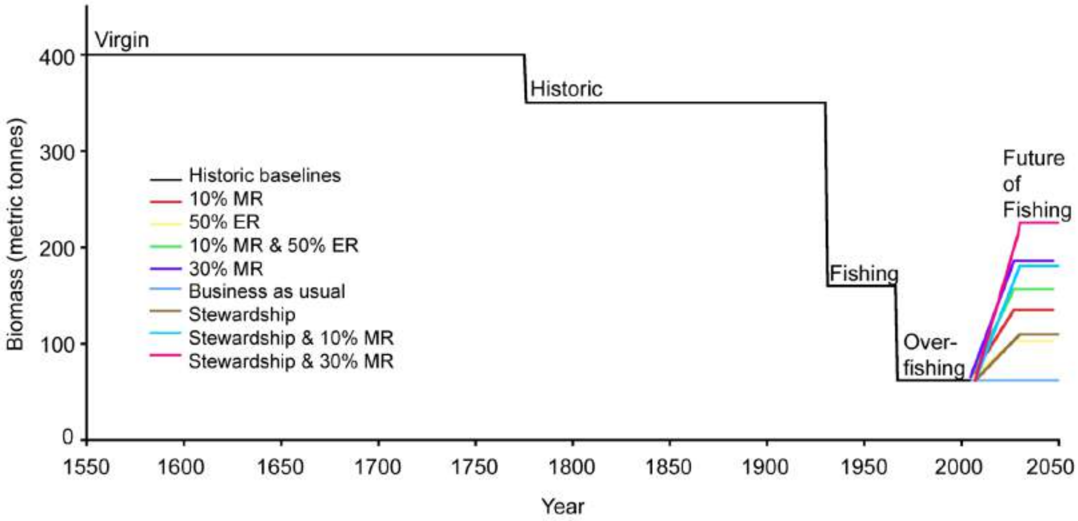


Figure 5. 4. Shifting baselines in lobster abundance in the Juan Fernández Archipelago. Calculated baselines for 1550-2008 and predictions for 'future of fishing' modeling scenarios (2008-2050).

period, the model did not accurately predict the frequency of variation in lobster catch, but was able to capture the magnitude of variation for the first part of the catch record and was able to predict the declining trend in lobster catch observed from 1967–1981 (Figure 5.5). The model does not accurately predict the last seven years (2001–2008) of the "over-fishing" period during which annual fluctuation in catch became highly variable immediately before the fishery was closed in 2004, and then rebounded in the following seasons. The model does however predict an increasing trend in lobster catch at the end of this period, corresponding to the observed catch record (Figure 5.5). The model is highly sensitive to the catchability term (*q*) as competing models without the term could not be optimised to run for the duration of the period.

Table 5. 3. Results of model selection analyses for the "Fishing" and "Over-fishing" Periods. RSS represents the residual sum of squares, K represents the number of parameters while AIC refers to the Aikaike Information Criterion value. Model with the lowest AIC value is indicated in bold. Δ*i* is the difference between the AIC value for each model and the model with the lowest AIC value (in bold).

'Fishing Period' (1930-1967)					
Model	Parameters	RSS	K	AIC	Δ*i*
A	ε, q	4283913	2	435.4	47.0
B	ε, P, q	1361955	3	395.0	6.6
C	α, ε, q	1290290	3	393.0	4.6
D	α, P, q	2023225	3	409.6	21.2
E	**α, ε, P, q**	**1079734**	**4**	**388.4**	**0**
F	q	3482531	1	425.7	37.3
G	α, q	1433071	2	394.9	6.5
H	P, q	3086683	2	423.3	34.9
'Over-fishing Period' (1967-2008)					
A	q	1301423	1	408.2	40.8
B	ε, q	1375611	2	412.4	44.9
C	α, q	913307	2	396.4	29.0
D	α, ε, q	636292	3	384.3	16.9
E	P, q	1301033	2	410.2	42.8
F	ε, P, q	1377988	3	414.4	47.0
G	α, P, q	1146888	3	407.3	39.8
H	**α, ε, P, q**	**391 972**	**4**	**367.4**	**0**

5.4.3 Model Prediction

The "future of fishing" model predicts the "business as usual" scenario to result in the lowest stock biomass at all times, peaking in 2027 at 111 tonnes, then declining slowly to 77 tonnes in 2050 (Figure 5.6A). The "stewardship and 30% MR" scenario resulted in the highest stock biomass which finishes at 225 tonnes in 2050 (Figure 5.6A). Scenarios that included the "stewardship" term maintained stock biomass at a relatively constant level after initial growth leveled off (Figure 5.6A). Scenarios that included the

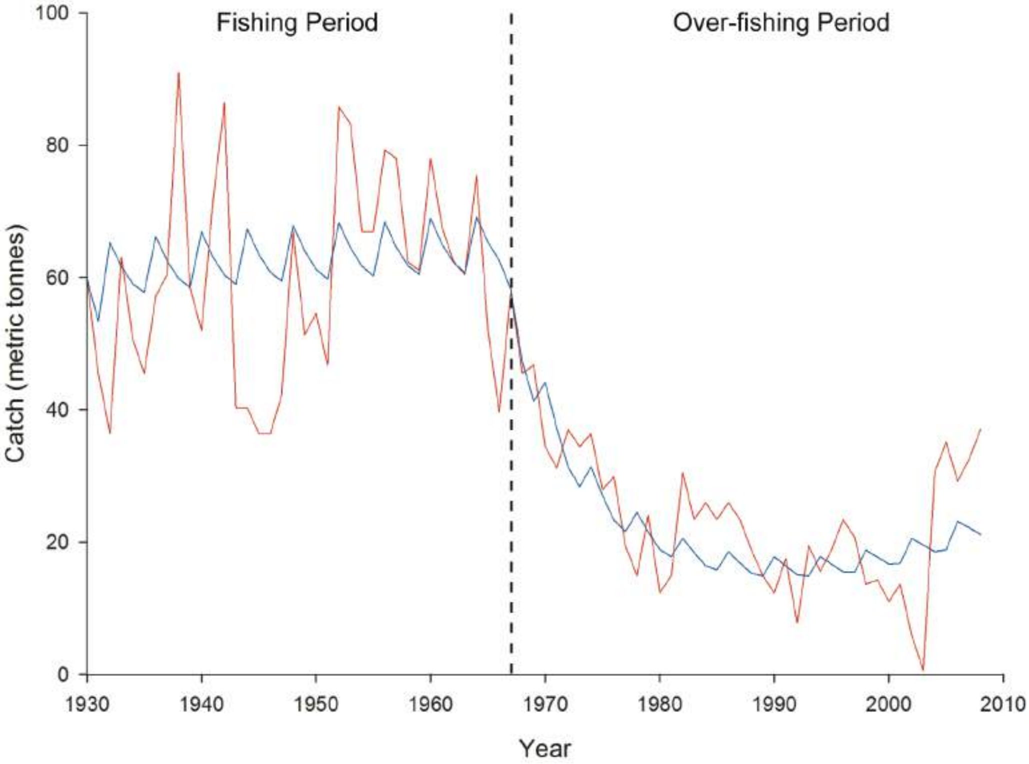


Figure 5. 5. Lobster catch for Islas Robinson Crusoe and Santa Clara. Estimated proportion of Chilean lobster (*Jasus frontalis*) fishery catch from Islas Robinson Crusoe and Santa Clara (red line) with predicted catch from bioeconomic fishery model (blue line) for the "Fishing Period" from 1930 – 1967 and the "Over-fishing Period" from 1967 - 2008.

"10% or 30% MR" term showed initial increases in stock biomass, slowly declining after approximately 15 years (Figure 5.6A). Scenarios that included the "50% ER" term showed a peak in stock biomass after approximately 15 years, finishing with a sharper decline (Figure 5.6A).

Scenarios that include the "50% ER" term predict the lowest catches throughout the first half of the scenario but finish with greater biomass than other scenarios (Figure 5.6B) and result in the lowest cumulative catch (Figure 5.6C). The "10% and 30% MR"

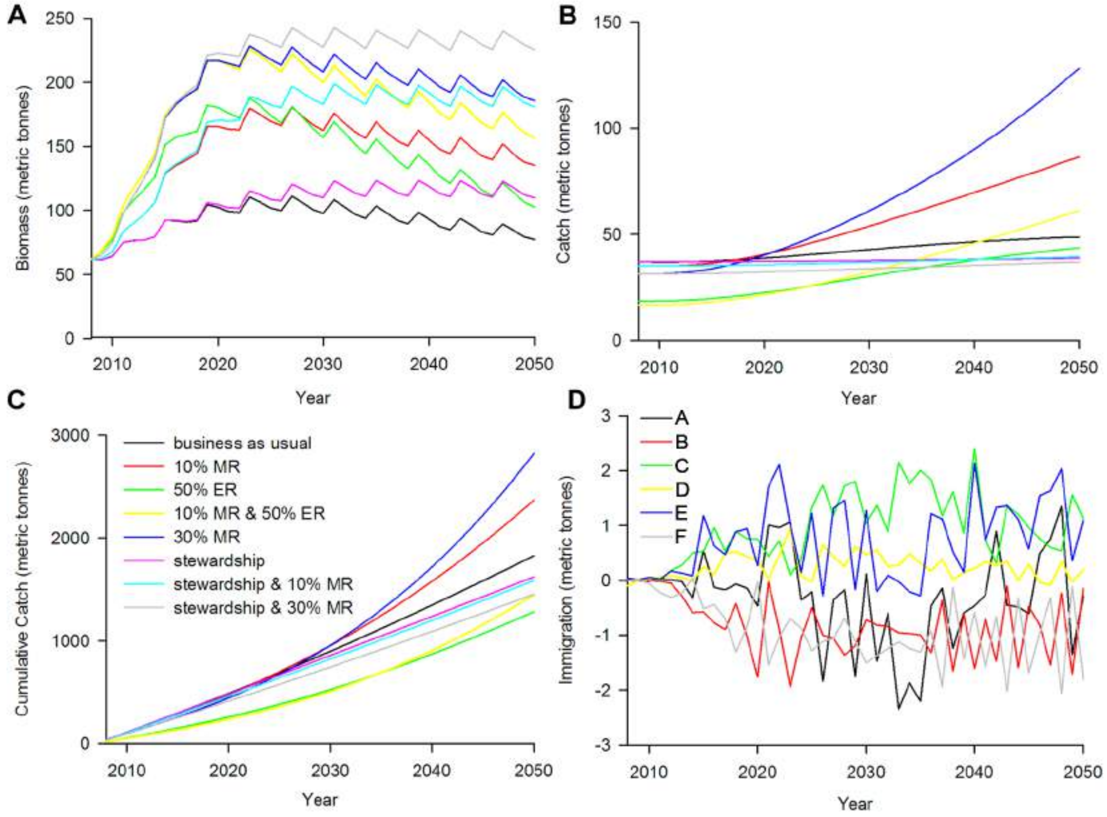


Figure 5.6. Predicted model results under different management scenarios. A - Predicted lobster biomass within the Juan Fernández archipelago for differing management and conservation strategies for the 'future of fishing' period from 2008 – 2050. B - Predicted model results for lobster catch in the Juan Fernández archipelago under different management and conservation strategies for the 'future of fishing' period from 2008 – 2050. C - Cumulative catch from 2008 -2050 predicted from the 'future of fishing' model for differing management scenarios. D - Predicted lobster spillover by the '30% MR' scenario from the 'future of fishing' model. Graph depicts change in population due to spillover in each region (measured as net change in population; positive values correspond to net immigration and negative values to net emigration). Regions A, B and F are closed to fishing with region A located in between regions B & F (see Figure 5.3 for map). Regions C and E are adjacent to areas closed to fishing, whereas region D does not share a boundary with a closed area (Figure 5.3).

scenarios predict the highest catch after approximately 12 years (Figure 5.6B) and the highest cumulative catch after approximately 20 years (Figure 5.6C). The “business as usual” scenario predicts relatively constant catch throughout the period, lower than scenarios with the “MR” term and higher than scenarios with the “stewardship” term (Figures 5.6B). Scenarios that include the “stewardship” term maintain catches at a constant level throughout the period (Figure 5.6B). All of the scenarios that include the “MR” and “ER” strategies show an exponential increase in cumulative catch at the end the period, while “business as usual” and “stewardship” strategies show a linear growth in cumulative catch (Figure 5.6C).

5.4.4 Trade-offs Between Catch and Stock Biomass

The “business as usual” and “stewardship” scenarios resulted in the highest catch initially due to absence of effort displacement, however the lobster population did not increase as quickly as in other scenarios (Figures 5.6A and B). The reduction in catch observed in scenarios that displace fishing effort through the use of MRs and ER allowed lobster biomass to increase, resulting in a greater rate of population growth. The “10% MR & 50% ER”, “30% MR” and “stewardship & 30% MR” scenarios resulted in the highest growth, but after 2019 the “stewardship & 30% MR” scenario maintained the largest biomass at ~235 tonnes while the “30% MR” and “10% MR & 50% ER” scenarios declined to ~200 tonnes and ~160 tonnes, respectively. The trade-off against the high biomass predicted to occur for the “stewardship & 30% MR” scenario is a reduced growth rate, resulting in less catch (Figures 5.6A and B). The “30% MR” scenario resulted in the highest catch in 2050 as well as the highest cumulative catch, while also maintaining a high biomass of ~200 tonnes (Figures 5.6A, B and C). The “business as usual” scenario resulted in the lowest biomass in 2050 and relatively constant catch throughout the scenario due to low growth associated with low stock biomass (Figures 5.6A and B).

5.4.5 Spillover Dynamics

Spillover predicted by the “30% MR” scenario from the “future of fishing” model resulted in net immigration of biomass for fished regions that shared a boundary with the MR (regions C & E) with average annual immigration values of 0.7 tonne and 1 tonne respectively (Figure 5.6D). Regions protected by the MR that also shared a boundary with fished regions (regions B & F) showed net emigration of biomass with average annual values of -0.8 tonnes and -0.9 tonne respectively (Figure 5.6D). The fished region that did not share a boundary with the MR (region D) exhibited less variability in spillover with an average immigration of 0.2 tonnes (Figure 5.6D). The region protected by the MR (region A) that shared boundaries with two regions also protected by the MRs was the most variable with an annual emigration average of -0.3 tonnes (Figure 5.6D).

5.5 Discussion

5.5.1 Factors Influencing Model Predictions

A number of different factors, ranging from fundamental aspects of lobster biology to aspects of fishers’ behaviour driven by economic necessity, may influence the outcomes of the different model scenarios. The high sensitivity of the model to the catchability (*q*) term suggests that changes to lobster trap technology resulting in greater catchability would have a substantial effect on the dynamics of the system. The small size of the human population, the size of the Juan Fernandez lobster fishery, and the isolation of the archipelago present a unique opportunity to explore these factors and how they might contribute to rebuilding or further decline of the endemic lobster stock.

5.5.1.1 Lobster population connectivity

Based on information about the dispersal of lobster larvae within the Juan Fernández Archipelago and the Islas Desventuradas (Dupré 2000; Rivera and Mujica 2004; Petrillo et al. 2005; Dupré and Guisado 1996), in my model I treated Islas Robinson Crusoe and Santa Clara as a closed system. Whereas evidence indicates limited exchange of larvae between Islas Robinson Crusoe - Santa Clara and Isla Alexander Selkirk (Rivera and Mujica 2004), the dynamics of source and sink populations between the Juan Fernández Archipelago and the Islas Desventuradas are unknown. The west wind drift runs from south to north, which suggests unidirectional flow from Juan Fernández to the Islas Desventuradas. Given this possibility, I suggest that the Robinson Crusoe-Santa Clara lobster fishery should be managed as a closed population.

5.5.1.2 Lobster movement

Knowledge of lobster movement is limited at the Juan Fernández Archipelago, with only reports of changes in depths that traps are set at during the fishing season to suggest changes in depth distributions (Arana and Vega 2000). In the absence of further information about movements of *J. frontalis*, I use information for *Jasus edwardsii* movement from New Zealand. *Jasus edwardsii* at the Cape Rodney to Okakari Point (Leigh) MR in northern New Zealand exhibited seasonal changes in depth distribution, sex ratio and size frequency which were related to moulting, reproductive and feeding cycles (MacDiarmid 1991). Additionally, Freeman et al. (2009) observed that *Jasus edwardsii* at Te Tapuwae o Rongokako MR in northeast New Zealand were more likely to be re-sighted on the same reef on which they were tagged and unlikely to cross muddy sediments between reefs. The configuration of the MR that protected 100% of one reef resulted in eight times greater abundance than another reef that was 91% protected by a MR (Freeman et al. 2009) indicating that *J. edwardsii* are more likely to “spillover” if MR boundaries occur over continuous rocky-reef habitat. Based on these findings I predict that lobster in the Juan Fernández Archipelago will respond positively to MR protection when such MRs are sited with due consideration of habitat requirements and natural barriers to dispersal. Further research quantifying larval dispersal patterns, recruitment, lobster movement and location of subtidal reefs and soft bottom at the Juan Fernández Archipelago would be valuable for MR design and model prediction.

5.5.1.3 Climate change, trophic interactions and disturbance

Recent climate change models predict that absolute fishery catch potential will increase slightly (0.5 to 5%) between 2005 and 2055 for the Juan Fernández Archipelago (Cheung et al. 2010). Trophic structure (and presumably trophic interactions) is not predicted to be affected by climate change as relative abundance of individuals at a given size is not strongly or consistently affected by temperature (Jennings and Brander 2010). New trophic interactions resulting from MR protection could result in higher abundances of lobster predators, however I do not suspect that this will be the case. As reported by fishers, the main predator of lobster is the octopus (Cabral et al. 2004), but this species is not targeted by fishers. I therefore do not expect the octopus to increase dramatically in abundance with MR protection and in addition, historic states of high lobster abundances in the presence of octopus and other predators have been documented (Walter 1776; Molina 1808; Albert 1898; Skottsberg 1956). Disturbance in the form of increased storm events arising from climate change (Jennings and Brander 2010) may impact lobster populations, although given their present depth distributions this seems unlikely. Increased natural disturbance such as storm events may however, contribute to a decrease in fishing activity as the small boats can venture out less often.

5.5.1.4 Illegal fishing

Estimates of illegal fishing activity are, by definition, hard to come by. While illegal fishing will inevitably slow (at low levels of poaching) or even prevent (at high levels) stock rebuilding regardless of the model scenario, measures have been initiated by fishers to prevent them (Ernst et al. 2010). I suggest that this is because the Juan Fernández Archipelago population and the lobster fishing community itself are both small, members of each are well known, and most families have a mutual interest in fishing. In addition, the geographic isolation of the archipelago offers protection against illegal fishing by “outsiders” which has been shown to break-down co-management institutions in other regions (Cudney-Bueno and Basurto 2009). As such, I doubt that illegal fishing activity will have a significant impact on the model scenario outcomes.

5.5.1.5 Heterogeneity in fishers’ responses

The response of fishers, in terms of modification of their own fishing behaviour, will contribute to stock rebuilding or ongoing decline (Wilen et al. 2002). Individual response among fishers with allocated property (fishing) rights may depend on a number of factors related to livelihood characteristics. It has been shown for fishers in mainland Chile that harvesting decisions may be related to mode of fishing and choices between leaving unfished stock to grow bigger in a subsequent year (e.g., dive fishers for the gastropod “loco”) versus taking stock now to permit immediate investment in new gear (e.g., generalist fishers using nets) (Gelcich et al. 2005; Gelcich et al. 2007). While the responses of the individual lobster fishers may vary according to such factors as personal financial pressure (size of mortgage repayments etc), the fact that all fishers are targeting one species, all are using the same gear, and the fishing community itself is small, leads me to suggest that the fishers' responses will be reasonably homogeneous.

5.5.2 Historic baselines at the Juan Fernández Archipelago

I have estimated a "virgin" lobster biomass of 400 tonnes. The current stock biomass, estimated at 60 tonnes (15% of virgin), is being maintained through an intensive fishery at the "over-fishing" baseline. There is evidence from New Zealand that historic baselines of lobster abundance can be achieved through the implementation of MRs, on small spatial scales and on timescales observable within a fishers' lifetime. At Te Tapuwae o Rongokako MR, the subtidal lobster population within the MR has reached density-dependence, such that foraging area has expanded to include a source of algal and invertebrate food sources located on the intertidal platform, an observation not witnessed at neighbouring unprotected locations (Lundquist and Pinkerton 2008). This observation is similar to the earliest (pre-exploitation) accounts on the Juan Fernández Archipelago (Walter 1776; Molina 1808) where lobster were reported in abundance in the intertidal zone, an indication of high densities in the subtidal region. The proportion of suitable habitat that is currently inhabited by lobster at Islas Robinson Crusoe and Santa Clara is a small fraction of historic accounts and FEK has confirmed greatest depletion of lobster with proximity to the port, such that the majority of the current catch is now made at the farthest reaches of the archipelago (Table 2). These verbal accounts are supported by reports of CPUE that are three times greater at Isla Alexander Selkirk and more than ten times greater at Islas Desventuradas (Arana and Vega 2000).

5.5.3 Rebuilding a Fishery

Suboptimal bioeconomic equilibrium is probable in small fisheries with little regulation (Beddington et al. 2007) and may be a legitimate management goal given that it is compatible with the sustainability of the resource (Ernst et al. 2010). Rebuilding stock biomass has the cost of catching less in the short term. The current “business as usual” management strategy is maintaining lobster biomass at an unproductive level, with catches at a historic low, is highly influenced by annual fluctuations, and has resulted in a reduced CPUE; tenfold less in comparison to the Islas Desventuradas (Arana and Vega 2000). The enforced closure of the fishery as occurred in 2004 is not a practical management strategy for fishers who already have an annual 4.5 month seasonal closure each year and a high dependence on the resource for their livelihood. However, the higher catches in the two years immediately after this enforced closure suggest that this type of action may be required again in the not too distant future as stock size will not be given a chance to rebuild.

My modeling results indicate that initial reductions in fishery catch caused by displacement of effort through the use of various management strategies can rebuild stock biomass to levels that can produce catches observed during the “fishing” period. The degree to which the stock biomass increases depends on the amount of effort reduction and/or area closed to the fishery. After approximately 10 and 15 years, catch and cumulative catch are predicted to be equal for both “10% MR” and “30% MR” scenarios respectively, in comparison to the “business as usual” scenario. The “stewardship & 30% MR” scenario rebuilds the stock biomass to the highest level, but provides significantly less catch to the fishery, indicating that this is a more conservation-minded strategy which trades off against economic gain. The “30% MR” scenario shows the greatest potential to increase both catch and cumulative catch while rebuilding the stock biomass to approximately 200 tonnes, half of the estimated “virgin” biomass. The degree to which stock biomass is rebuilt will depend on the level of "stewardship" displayed by fishers. Any poaching of the MR will obviously impact its performance to rebuild the lobster population.

It has been suggested that it is for the benefit of the Juan Fernández fishing community that a TAC has not been implemented, as it would disrupt a system of informal traditional tenure (Ernst et al. 2010). In the absence of TAC, MRs provide an insurance policy for the stock, to ensure that a portion will remain unfished and intact. While a dynamic approach (i.e., rotating the location of the MR) may benefit some trophic groups (Game et al. 2009), I suggest that in the present case where the focus is on a "sessile" species, or at least a species with low mobility, a static MR approach will be more beneficial in line with findings elsewhere (Kelly et al. 2000; Davidson et al. 2002; Shears et al. 2006; Pande et al. 2008). This approach also has the benefit of being easier to delineate (on maps and with coastline markers or buoys) and easier to enforce. The proposed MR location was sited by fishermen as it will displace minimal fishing effort (most depleted of lobster) and so that it can be observed by village members.

Despite the obvious long-term (sustainability) benefits of a co-management and stewardship strategy, implementing fisheries management practices where this phenomenon is observed is often the real challenge (Hardin 1968; Rosenberg 2007). Increasing ownership and implementing community-based co-management of the fishery catch has been shown to increase compliance, promote conservation values and transfer the burden of management and enforcement by using a bottom-up approach (Gelcich et al. 2005; Castilla and Fernandez 1998; Gelcich et al. 2008). My conversations with fishers indicated a sense of disparity in the historically low catches in comparison to higher catches from the "good old days", which has been shown to foster a greater willingness to change existing practices (Gelcich et al. 2005). Recent reports (Ernst et al. 2010) indicate that fishermen have insisted on the need for a regular presence of the fisheries authority on the islands to improve the quality of landing statistics and the enforcement of size regulations. As a result, a voluntary logbook program has been started to record spatial CPUE data, which is a good indicator of stock abundance (Ernst et al. 2010). Following my conversations with fishers where I introduced the idea of MRs to many of them for the first time (Eddy et al. 2008), it has been reported that the lobster fishers' syndicate is trying to create a MR extending to 10 nautical miles around the islands with the ultimate goal of excluding mainland-based fishing fleets (Ernst et al. 2010). Based on recent participation and demand for inclusion in management decisions by lobster fishers to employ regulatory, monitoring and conservation initiatives I believe that my proposed co-management strategy to determine the level to which stock biomass is rebuilt is realistic and compliance would be high. The isolation of the Archipelago makes it unlikely that "roving bandits" from other fishing communities (Cudney-Bueno and Basurto 2009) pose a threat.

5.5.4 The Use of FEK and Historic Sources

My approach, employing the bioeconomic fishery model for the long time period I examined has strength in its ability to place current biomass stock in the context of virgin biomass. Given that estimates of lobster biomass throughout time are patchy, often qualitative and not spatially resolved, FEK and historic sources permit investigation of the current state of resource exploitation. I do not seek to estimate how much lobster can be taken today without causing the population to collapse, that is the realm of stock assessments. My aim is to show how trading off some catch today will not only provide greater landings and higher CPUE in the future, but also provide a whole suite of conservation and ecosystem-based management achievements through MR implementation, while giving control and responsibility of the resource to the fishers. The voluntary CPUE logbook program that is now in place (Ernst et al. 2010) will provide an accurate, spatially resolved indicator of abundance to allow for better informed management and conservation decisions in the future.

Chapter 6

General Discussion

While I have written each data chapter of this thesis to be self-contained for publication purposes, I have centred my research around a few common themes. The major aim of my PhD thesis is to provide an understanding of how using MRs for conservation and management tools impacts both the marine communities they protect but also the people who use coastal resources. I have utilised a number of different techniques and methods to address the fundamental question of this thesis: "How do coastal resource use and MRs affect marine animal populations?" These techniques are useful across a wide range of scenarios and are not limited to the case studies that I have applied them to. I now address the overall themes that have been presented across chapters of my thesis in order to provide broader analysis.

6.1 Reef Fish Assemblages in New Zealand

Throughout chapters 2, 3 and 4, I have employed the use of underwater surveys to monitor size and abundance of reef fishes in the shallow coastal waters of New Zealand. The locations of these surveys have varied from the temperate waters of the Cook Strait (Kapiti and Taputeranga MRs), located between the North and South Islands and the subtropical waters of New Zealand's northernmost marine environment at the Kermadec Islands. Along with the biogeographic gradient among these sites, there is also a gradient of human exploitation of marine resources. While some fishing for shallow coastal species and tunas has taken place throughout history, this has had minimal impacts on the marine ecosystem, which has been identified by the Census of Marine Life as one of 20 remaining pristine marine ecosystems left in the world. Conversely, the Cook Strait region of the North Island has experienced approximately

800 years of human exploitation by Maori peoples and over 100 years by Europeans and their decedents.

The observation of planktivores accounting for half of the biomass of all reef fishes at the Kermadec Islands (chapter 2) differs in comparison to New Zealand's temperate waters. Observations from Taputeranga MR in central New Zealand (chapter 4) indicate that biomass of reef fishes in the shallow subtidal is dominated by invertebrate feeders. Piscivores and herbivores account for a smaller amount of biomass, while planktivores make up the least amount of biomass of all the trophic groups. This indicates that the food sources that are utilised by reef fishes in temperate New Zealand waters are markedly different from those used in the subtropical waters of the Kermadec Islands. This is likely attributed to the large biogeographic distance between the two locations and the availability of food items in two very differently structured ecosystems. It should also be noted that exploitation history may be a contributing factor in the differential organisation of reef fish assemblages given the different influences of human coastal resource use on marine ecosystems among locations. This is a situation where MRs can provide a comparison between exploited and unfished ecosystems. At the Kapiti MR, it does not appear that planktivores are increasing in any great abundance or biomass in comparison to fished control sites. At Kapiti MR, herbivores and invertebrate feeders are responding the most positively to MR protection. Hence, it appears that not only the fish fauna, but also the trophic structure differs between the subtropical waters of the Kermadecs and the temperate waters of central New Zealand.

6.2 Lobsters, their Ecosystem Roles and Lobster Fisheries

At areas surrounding the Taputeranga MR, New Zealand and at the Juan Fernández Archipelago in Chile, lobster (*Jasus edwardsii and J. frontalis* respectively) is the principal economic resource. At both of these locations, lobster stock biomass is being maintained at a fraction of its former abundance through exploitation by fisheries. At the Taputeranga MR, my results show that the level of lobster abundance in the ecosystem influences its keystone role, a measure of the importance of its role in ecosystem organisation. I suggest that this is also the case at the Juan Fernández Archipelago as the biology and foraging behaviour of *Jasus frontalis* is very similar to *J. edwardsii*. It should be noted that the temperate benthic community structure differs quite dramatically between the Taputeranga MR and the Juan Fernández Archipelago as there is an absence of kelp at the Juan Fernández Archipelago. Although kelp is only a minor diet component of lobsters in the Taputeranga MR, it accounts for the most biomass of any trophic group at Taputeranga and therefore, primary productivity differs markedly between these communities.

Co-management of lobster in NZ has resulted in many benefits for the lobster fishery in that it has produced more predictable catches through the use of recruitment studies and complex scientific population modeling. This has resulted in a situation that has led to voluntary reductions in catch by fishers, who make recommendations about TAC to the governing body, the Ministry of Fisheries (Yandle 2006). The management structure and the nature of the lobster fishery in the Juan Fernández Archipelago is very different from that of New Zealand. The artisanal low-tech fishery operating on the Juan Fernández Archipelago does not employ the use of TAC or quotas to manage the amount of lobster taken, however is governed by a complex system of informal regulations regarding trap placement, which in turn affects CPUE of individual fishers. The commonality between the two fisheries lies in the fact that both fisheries have limited entry, which means that each fisher who holds rights to fish has a share in the overall stock and therefore a common interest to maintain the health and productivity of the stock. With respect to the co-management analysis of the Juan Fernández Archipelago lobster fishery, it is more appropriate to look at the co-management situation within Chile which occurs for other coastal marine resources which has been summarised in Chapter 5.

6.3 Historic Baselines and Future Predictions

In order to understand how marine animal populations have changed throughout time, I have compared current abundances to historic levels, a field that is known as historical ecology, which has been gaining popularity since the seminal paper published by Jackson et al. (2001). By placing current abundances or ecosystem state on a time scale beginning with the start of human exploitation of coastal resources, an understanding of marine animal populations in their pristine state can be compared to present states. In the absence of quantitative data about changes in marine animal populations throughout history, alternative methods are required. By using historic accounts of abundance and FEK from the Juan Fernández Archipelago, Chile it was determined that the current abundance of lobster is a fraction of 'pristine' levels. The approach also revealed that the Taputeranga MR ecosystem is functionally different in past and future states when lobster abundances are estimated and predicted to be much higher. In both cases, the potential of MRs to return stock biomasses and ecosystem states to those observed in historic times was assessed through the use of models.

The use of models in ecology can provide insight into complex systems and their behaviour that otherwise might not be apparent or intuitive. Ecosystem models are a tool that can be used to understand how marine animal populations are connected and how exploitation and climate-related events impact not only singles species, but entire ecosystems. At the Taputeranga MR, this approach has identified species that play an important role in ecosystem functioning as well as provided insight into the recovery of exploited species and their impacts on other non-targeted species that may not be apparent otherwise. Ecosystem modeling has recently been applied to management scenarios following the failure of single-species assessments to represent ecosystem dynamics which has resulted in the widespread call for the ecosystem-based management approach to exploitation of coastal resources (Browman et al. 2004). The use of a bioeconomic fishery model evaluated different management strategies such as the use of MRs, to predict their abilities to rebuild over-exploited lobster stocks with consideration of economic implications at the Juan Fernández Archipelago. This approach was used as a decision-based tool for the management of coastal resources to determine tradeoffs between catch and conservation goals. In both the case of the ecosystem model and the bioeconomic fishery model, they have been applied to describe the past, understand the present and predict the future.

6.4 The Use of MRs and Implications for Coastal Resource Use

Overall, my thesis has provided a synthesis of methods for investigating the use of MRs for conservation and management of coastal resources. The use of timed underwater counts documented fish assemblages and trophic structure at the subtropical, pristine ecosystem that is protected by New Zealand’s largest MR at the Kermadec Islands. Documenting the response of reef fishes to 18 years of MR protection at Kapiti Island in New Zealand using underwater transect counts has indicated that butterfish biomass has increased tenfold in the last 10 years and previously targeted species have increased threefold. Modeling indicates ecosystem functioning is different on the Wellington south coast following 70 years of marine resource exploitation, however is predicted to return to a more historic state within the Taputeranga MR following MR implementation. Bioeconomic modeling showed that at the Juan Fernández Archipelago, lobster stocks have been depleted to a small fraction in comparison to pristine levels and could be rebuilt to more historic levels through the use of a MR.

When considering the potential of MRs for conservation and management tools, incorporating stakeholder groups in the process is imperative to evaluate the potential success of the MR. Without the support of stakeholder groups, the design, implementation and compliance of a MR is unlikely to be successful. By incorporating stakeholder groups from the onset of the MR process, not only can potential issues be indentified, but also knowledge about local conditions can be learned. This is one of the key messages to consider when evaluating management tools; they are about managing people first and resources second. Without consideration of the social aspect of management tools, it is likely that their application from theory to practice will fail. Fishers in general are protective of their fishing grounds and need to be convinced that there is a good reason for forfeiting an area that they have traditionally fished in order to achieve some conservation goal. If they can be convinced about benefits that implementing a MR would have for a marine ecosystem and species that are fished then it might be possible that they will endorse the proposal. My presentation to the fishers of the Juan Fernández Archipelago about the potential of a MR and its impacts on the lobster fishery was initially met with scepticism and criticism. It was only after a long question and answer period about how MRs have been shown impact both lobster and lobster fisheries in New Zealand, did they start to become onside with the issue. Now, fours year later, there is a movement by the lobster fishers themselves to establish a MR at the Juan Fernández Archipelago.

The impact of MRs on marine ecosystems varies depending on geographical location, at which spatial and temporal responses by species are highly variable. My observations from the two central New Zealand MRs that I studied in which were compared to neighbouring sites subject to exploitation suggest that given specific criteria are met, some general outcomes can be expected. The criteria relate to: compliance of the MR; the size and location of the MR are suitable to protect a large enough area of habitat that is used by exploited species; sufficient recruitment of exploited species into the MR occurs. Given these criteria are met, it can be predicted that previously exploited species will increase in abundance, biomass and average size as a function of their biology. For example, highly mobile species such as kingfish (*Seriola lalandi*) are unlikely to positively respond to protection at the Kapiti and Tauputeranga MRs, however benefit from protection at the much larger Kermadec Islands MR. Species such as paua (*Haliotis australis and H. iris*), blue cod (*Parapercis colias*), butterfish *(Odax pullus*), blue moki (*Latridopsis ciliaris*) and lobster *(Jasus edwardsii*), which are less migratory have been shown to respond positively in smaller MRs in New Zealand. Therefore, response of exploited marine species to MR protection is a function of their biology as well as the size and location of the MR.

The response of an entire marine ecosystem to MR protection is a bit harder to generalise, however it can be stated that the degree of ecosystem degradation is an important factor. Some ecosystems have shown large structural changes in areas with a history of intensive exploitation, such as the Adriatic Sea (Coll et al. 2008b) or in areas where large-scale changes in environmental conditions have taken place such as the eutrophication of the Baltic Sea, which led to a regime shift in the associated marine ecosystem (Tomczak et al. 2009). Conversely, marine ecosystems that have experienced less intensive exploitation and changes in environmental conditions are more likely to respond positively to MR protection resulting in a marine community more similar to that of pre-exploitation.

Appendices

Appendix A

Eddy TD (2011) Recent observations of reef fishes at the Kermadec Islands Marine Reserve, New Zealand. New Zealand Journal of Marine and Freshwater Research. (Front page only, based on Chapter 2).

New Zealand Journal of Marine and Freshwater Research
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Recent observations of reef fishes at the Kermadec Islands Marine Reserve, New Zealand

TD Eddy*

Centre for Marine Environmental and Economic Research, School of Biological Sciences, Victoria University of Wellington, Wellington, New Zealand

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The Kermadec Islands have been identified as one of the few remaining pristine marine ecosystems left in the world. The Kermadec Islands Marine Reserve (MR) is the largest in New Zealand protecting species endemic to the archipelago and species not found elsewhere within the country. Reef fishes were surveyed for size and abundance at three sites around Raoul Island and the Meyer Islets and biomasses of trophic groups were calculated. Planktivores dominated trophic group abundance at all three sites. This research represents the first observations of all trophic groups of reef fishes since implementation of the Kermadec Islands MR in 1990.

Keywords: Kermadec Islands; marine reserve; reef fish; subtropical; trophic

Introduction

The isolated Kermadec Islands (29–31°S, 178°W), located 750 km northeast of Cape Reinga, represent the only true subtropical marine habitat in New Zealand. The volcanic Kermadec Island archipelago is composed of four main island groups; Raoul Island and surrounding Herald Islets in the north, Macauley and Haszard Islands, Curtis and Cheeseman Islands and the southernmost island group of L'Esperance and Havre Rocks (Fig. 1). Located between New Zealand (34°S) and Tonga (21°S), the Kermadec Islands harbour a mix of temperate and tropical species (Schiel et al. 1986; Francis et al. 1987; Francis 1991; Cole et al. 1992; Francis 1993; Brook 1998, 1999; Cole 2001; Gardner et al. 2006; Wicks et al. 2010). Sea surface temperature varies from 18° to 24°C seasonally (Francis et al. 1987). While corals are present, both coral reefs and macroalgal stands are absent at the Kermadec Islands (Schiel et al. 1986; Brook 1999).

During the time of initial studies by Francis et al. (1987) in 1984 and 1985, long-line fishing pressure in New Zealand was beginning to expand geographically following declines of hapuku (*Polyprion oxygeneios*) and bass (*Polyprion americanus*) stocks with fishing trips planned to the Kermadecs. Subsequently Francis submitted an application (1985) for a marine reserve (MR) to protect New Zealand's only subtropical marine ecosystem and the Kermadec Islands MR was designated in 1990. It is New Zealand's largest MR at 748,000 ha and extends 22 km seaward from all four island groups (Fig. 1). The Kermadec Islands MR protects species endemic to the Kermadec Islands as well as highly targeted commercial species in a region where recruitment occurs locally for some species (Francis et al. 1987). The Kermadec Islands have recently been identified by the

*Email: tylereddy@gmail.com

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Appendix B

Eddy TD, Gardner JPA, Pérez-Matus A (2010) Applying fishers' ecological knowledge to construct past and future lobster stocks in the Juan Fernández Archipelago, Chile. PLoS ONE. (Front page only; based on Chapter 5)

OPEN ACCESS Freely available online

PLoS one

Applying Fishers' Ecological Knowledge to Construct Past and Future Lobster Stocks in the Juan Fernández Archipelago, Chile

Tyler D. Eddy[1]*, **Jonathan P. A. Gardner**[1], **Alejandro Pérez-Matus**[2]

1 School of Biological Sciences, Centre for Marine Environmental and Economic Research, Victoria University of Wellington, Wellington, New Zealand, 2 Victoria University Coastal Ecology Laboratory, School of Biological Sciences, Victoria University of Wellington, Wellington, New Zealand

Abstract

Over-exploited fisheries are a common feature of the modern world and a range of solutions including area closures (marine reserves; MRs), effort reduction, gear changes, ecosystem-based management, incentives and co-management have been suggested as techniques to rebuild over-fished populations. Historic accounts of lobster (*Jasus frontalis*) on the Chilean Juan Fernández Archipelago indicate a high abundance at all depths (intertidal to approximately 165 m), but presently lobsters are found almost exclusively in deeper regions of their natural distribution. Fishers' ecological knowledge (FEK) tells a story of serial depletion in lobster abundance at fishing grounds located closest to the fishing port with an associated decline in catch per unit effort (CPUE) throughout recent history. We have re-constructed baselines of lobster biomass throughout human history on the archipelago using historic data, the fishery catch record and FEK to permit examination of the potential effects of MRs, effort reduction and co-management (stewardship of catch) to restore stocks. We employed a bioeconomic model using FEK, fishery catch and effort data, underwater survey information, predicted population growth and response to MR protection (no-take) to explore different management strategies and their trade-offs to restore stocks and improve catches. Our findings indicate that increased stewardship of catch coupled with 30% area closure (MR) provides the best option to reconstruct historic baselines. Based on model predictions, continued exploitation under the current management scheme is highly influenced by annual fluctuations and unsustainable. We propose a community-based co-management program to implement a MR in order to rebuild the lobster population while also providing conservation protection for marine species endemic to the Archipelago.

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* E-mail: tylereddy@gmail.com

Introduction

As a response to reports of declining and unsustainable fisheries worldwide [1–5] there has been much debate among conservationists, fisheries biologists and fisheries managers [6] about the best means to balance sustainable exploitation with conservation of biodiversity and ecosystems. Proposed solutions include, but are not limited to, ecosystem-based management, MRs and other forms of fishery closures, incentives, co-management, total allowable catch (TAC) and individual transferable quotas, reductions in fishing fleet capacity and changes in gear regulations [7–9]. Elsewhere it has been suggested that the tools for effective management of fish stocks are already available and that fishery science is sound, but that recommended harvest limits are rarely implemented as policy because of lobbying by stakeholders [10]. We examine co-management strategies in Chile where the stakeholder group that most often objects to fisheries regulations, fishers, has taken a major role in the management of their livelihood. In the absence of information about the response of lobster to MRs in Chile, we examine the potential of MRs for fisheries management in the Juan Fernández Archipelago using observations from New Zealand. We then investigate the effects co-management, MRs and traditional fisheries management tools for their effectiveness to rebuild an overexploited Chilean lobster (*Jasus frontalis*) fishery as well as promote conservation values and ecosystem protection.

Marine Reserves and Co-Management

In Chile and New Zealand, MRs are implemented for conservation purposes, but may produce indirect benefits for fisheries because they have been shown to increase the size, abundance and biomass of many fished species, including the New Zealand lobster, *Jasus edwardsii* [11–14], to safe-guard against fishery-associated handling disease [15], and to increase population-specific egg production rates because larger lobsters produce disproportionately more eggs than smaller lobsters [11]. However, while the area of the MR may benefit from a reduction in fishing pressure, adjacent areas may not. For example, the implementation of a MR often displaces fishing effort, resulting in greater fishing effort per unit area outside the MR [16–19]. A concern

Appendix C

Eddy TD (2009) New Zealand’s Largest and Most Isolated Marine Reserve: the subtropical Kermadec Islands. JMBA Global Marine Environment.

NEW ZEALAND'S LARGEST AND MOST ISOLATED MARINE RESERVE:
the subtropical Kermadec Islands

FIGURE 3. Diversity of echinoderms: crown-of-thorns starfish (*Acanthaster planci*) and feather star (Photograph courtesy of Pete Mesley).

The volcanic Kermadec Islands are home to New Zealand's northernmost land mass, located between 29 and 32° south latitudes and 177 and 179° west longitudes. The archipelago is composed of four island groups, located 750 km north-east of New Zealand along the Kermadec Trench which extends from the East Cape. The Kermadec Islands are found approximately halfway between temperate New Zealand waters and tropical Tonga Island waters. Raoul Island belongs to the northernmost group, is the largest of all the islands in the archipelago at 2938 ha and is surrounded by the smaller Meyer Islands and Herald Islets. The next island group to the south-west is the Macauley and Haszard Island group, followed by the Curtis and Cheeseman Island group with L'Esperence Rock and Havre Rock as the most southerly group in the Kermadec Island chain (figure 1).

MARINE RESERVES IN NEW ZEALAND

The first marine reserve (MR) in New Zealand was created at Leigh in 1975, located north of Auckland in the Hauraki Gulf and was named the Cape Rodney–Okakari Point MR (also known as the Goat Island MR). Within 10 years of protection, major changes in community structure within the reserve were noted as the area had previously been exploited by over-fishing. Prior to protection, the benthic community was largely dominated by urchin (*Evechinus chloroticus*) barrens, a grazer of macroalgal species. After the area was protected from fishing, abundance of the highly targeted snapper (*Pagrus auratus*) increased, which is a natural predator of sea urchins. The growth of the snapper population decreased urchin abundance, which allowed previously highly-grazed macroalgae species to return in larger numbers. Macroalgae stands provide habitat and structure for many invertebrate and fish species, thus resulting in a large increase in biodiversity and major changes in biomass within the MR. Trophic cascades such as this have been documented at other MRs in New Zealand, often occurring within a few years of protection (Shears & Babcock, 2002, 2003).

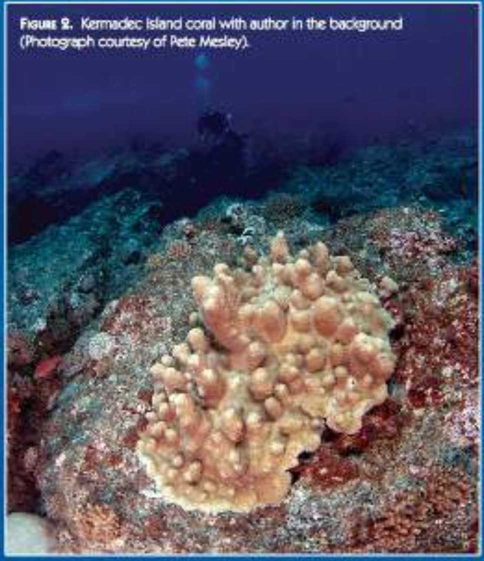

FIGURE 2. Kermadec Island coral with author in the background (Photograph courtesy of Pete Mesley).

The Kermadec MR was created in 1990 and is New Zealand's largest at 748,000 ha. The MR protects the four main island groups, extending to 12 nautical miles seaward. Protected marine habitat ranges from the high intertidal zone to the shallow coastal waters surrounding the islands dropping to 40 m, to deeper waters of depths greater than 3000 m. Marine reserve law in New Zealand prohibits fishing, removal of any living matter or disturbing marine communities within the reserve.

The Kermadec MR (figure 1) protects New Zealand's only true subtropical reef habitat with sea-surface temperatures that range from 18° to 24°C annually. The marine community is a unique mix of tropical, subtropical and temperate species. Benthic primary producers are represented by scleractinian corals (figure 2) which are found at shallow depths as well as foliose, filamentous and encrusting red algae. Notably absent are fucalean and laminarian algae. Grazing invertebrates are represented by echinoderms such as the crown-of-thorns (*Acanthaster planci*, figure 3) and gastropods such as the endemic giant limpet (*Patella kermadecensis*). Herbivorous fish are represented by the grey and caramel drummer (*Kyphosus bigibbus* and *Girella fimbriata*) and the Pacific gregory (*Stegastes fasciolatus*). Demoiselle (*Chromis dispilus*; figure 4), blue maomao (*Scorpis violaceus*), orange wrasse (*Pseudolabrus luculentus*) and mimic blenny (*Plagiotremus tapeinosoma*) make up the balance for the most observed species. Larger fish species are represented by the spotted-black grouper (*Epinephelus daemelii*), kingfish (*Seriola lalandi*), northern kahawai (*Arripis xylabion*) and Galapagos shark (*Carcharhinus galapagensis*; figure 5 background). It has been determined that the fish fauna of the Kermadec Islands is more similar to Lord Howe Island than to northern New Zealand.

CONDUCTING RESEARCH AT THE KERMADEC ISLANDS

On 22 March 2008, I joined the vessel 'Spirit of Enderby' operated by Heritage Expeditions in Auckland, New Zealand destined for the Kermadec Islands. Research expeditions to these islands are infrequent due to their isolation and narrow weather windows with regular cyclone activity. I was fortunate to have been awarded a scholarship by the Enderby Trust to join the cruise with tourists interested in different facets of the Island's history, birdlife and underwater environs. The research that I conducted while at the islands will be used for my PhD thesis at Victoria University of Wellington titled: 'Marine Reserves as Conservation and Management Tools in New Zealand,' supervised by Dr Jonathan Gardner and Dr James Bell.

Survey sites were highly determined by the prevailing wind, which was from the east for most of our time and meant that Denham Bay was the most dived site during the trip (figure 1). The physical structure of Denham Bay is gently sloping from the intertidal to subtidal zone and characterized by large boulders, rocks, cobble and sand. One survey was conducted at the Meyer Islands, characterized by more vertical walls with habitat features such as caves, overhangs and crevices.

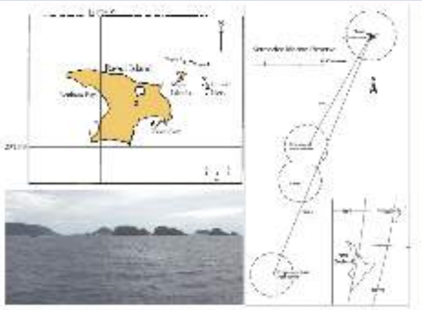

Figure 1. Map of survey locations (a, b & c). Kermadec Marine Reserve boundaries (dotted lines). Photograph of Raoul Island, Meyer Islands and Herald Islets. Kermadec Marine Reserve map courtesy of the Department of Conservation (Photograph by Tyler Eddy).

Surveys of benthic and pelagic fish species for size and abundance were conducted using a two minute timed count technique. The survey was designed to sample a range of depths and habitats in order to better understand how much variability in the fish distribution size and abundance can be explained by these factors. The body of research pertaining to the unique fish community found at the Kermadec Islands is relatively small, with few published studies (see Schiel *et al.*, 1986; Francis *et al.*, 1987; Cole *et al.*, 1992, 2001; Francis *et al.*, 1999). This research represents the first survey conducted for all species of fish since the opening of the Kermadec MR in 1990 and one more snapshot of the distribution of fish in this isolated location.

During the five surveys 38 species of fish were recorded for size and abundance and three further species were sighted for a total of 41 observed fish species. There have been more than 100 species of fish described at the Kermadec Islands and it has been determined that greater sampling effort and more survey sites throughout the archipelago are required in order to increase the number of species observed. Preliminary results indicate that fish assemblages are significantly different between Denham Bay and Meyer sites. Further analysis is underway to determine the driving factors that are responsible for differences.

CONSERVATION IMPLICATIONS

Based on estimates of population sizes and densities of marine communities at the Kermadec Islands, it has been suggested that recruitment may be a limiting factor in a region that is highly isolated, subject to strong wave and storm energy manifested through cyclones as well as the relatively small amount of suitable rocky-reef habitat available due to water depths dropping to as deep as 3000 m within the marine reserve. It has also been noted that there are no major currents flowing to the Kermadec Islands, which may explain why major coral and macroalgal groups that are expected to be able to survive in such conditions are absent.

For these reasons, this ecosystem is exceptionally fragile to anthropogenic factors such as invasive species introduced from boat hulls or bilge water as well as fishing. If the population does in fact rely on self-recruitment, fishing pressure could have dramatic impacts on the community structure of fish throughout the archipelago. Particularly susceptible is the spotted black grouper (*Epinephelus daemelii*) which is female until it grows to 100–110 cm and changes sex. Fishing of larger individuals could potentially remove all the males from the population. There are also endemic invertebrate (giant limpet; *Patella kermadecensis*) and fish species (Kermadec scalyfin; *Parma kermadecensis*, *Ocosia apia* and possibly *Gymnothorax griffini*) which merit special conservation priorities.

The Kermadec MR protects a unique marine community that appears to be extremely susceptible to human influence. A major reason why the Kermadec Islands are still home to spotted black grouper, Galapagos reef sharks, endemic giant limpet and Kermadec scalyfin is the geographical isolation of the islands which were for a large part of history not regular fishing grounds. The MR that is in place today protects a unique biological community not found elsewhere in the world.

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Tyler D. Eddy
Centre for Marine Environmental and Economic Research,
School of Biological Sciences, PO Box 600
Victoria University of Wellington, New Zealand.
E-mail: *tyler.eddy@vuw.ac.nz*

BACKGROUND: Figure 5 Galapagos reef sharks (*Carcharhinus galapagensis*) (Photograph by Tyler Eddy).

Figure 4. Demoiselle (*Chromis dispilus*), feather stars and gorgonians at Meyer Islands (Photograph by Tyler Eddy).

Appendix D

Eddy TD, Ramírez F, Perez-Matus A (2008) Oceanic Islands: The Chilean Juan Fernández Archipelago. From Natural Observations to Management Challenges. JMBA Global Marine Environment.

FIGURE 1. Aerial view Robinson Crusoe Island (from north to south). Photograph by Fabian Ramirez.

OCEANIC ISLANDS:

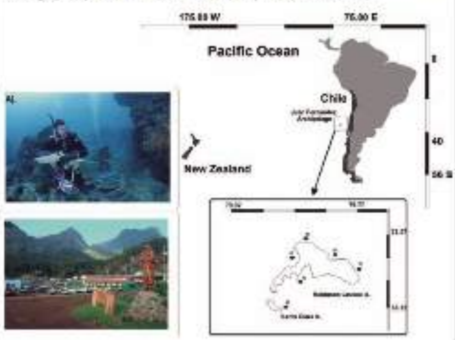
The Chilean Juan Fernández Archipelago. From Natural Observations to Management Challenges.

Early explorers to visit the Juan Fernández Archipelago in the 1700's found that lobsters were "in such abundance near the water's edge (of Isla Robinson Crusoe) that the boat-hooks often struck into them, in putting the boats to and from the shore" (Walter 1776) and were "found in such quantities that the fishermen have no other trouble to take them, than to strew a little meat upon the shore, and when they come to devour this bait, as they do in immense numbers, to turn them on their backs with a stick" (Molina 1808). We did not find such a plethora of the Islands' principal economic resource, the lobster.

We traveled to the Juan Fernández Archipelago in September 2007 to conduct research in subtidal habitats as part of our study for the Ph.D programme at the Victoria University of Wellington, funded by Education New Zealand Study abroad grant, and to undertake species collection for a global taxonomic review of ichthyofauna genres in collaboration with the fish research department at Te-Papa Museum New Zealand.

Oceanic Islands are those situated far from all continents and the Juan Fernández Archipelago fits perfectly into this category (Oliva & Castilla, 1987). Composed of three islands, the Juan Fernández Archipelago was discovered by the Portuguese sailor Joao Fernández in 1574. At that time, these Islands served as a source of provisions and a refuge for pirates. The Archipelago is the setting used by Daniel Defoe to write his best selling book named "Robinson Crusoe" inspired by the Scottish sailor, Alexander Selkirk who lived there for four years in solitude. Formerly named "más a tierra" (close from shore) and "más a fuera" (far from shore) now "Robinson Crusoe", "Santa Clara", along with "Alejandro Selkirk", coined for tourism, are located in the south Pacific (33° 37'S – 78° 51' W) 680 km off the port of San Antonio, Chile. These islands are of scientific interest for several different disciplines in biology and social sciences. These islands represent natural laboratories as they are an important source of speciation which taxonomically can be explained by the high degree of endemism for both terrestrial and marine groups (scientific studies, for example, have revealed 32, 67, 70, 15, 23, and 45 per cent of endemism for macroalgae, anthozoans, molluscs, decapods, crustaceans, echinoderms and reef fishes respectively Rozbaczylo & Castilla, 1987, Santelices, 1992; Pequeño & Lamilla, 2000). Biogeographic studies of Juan Fernández Archipelago have yielded a greater affinity with distant continents then

FIGURE 2. Map Showing Juan Fernández in respect to continental mainland Chile and New Zealand, the sampling sites (study sites a) Bahía Cumberland, b) El Frances, c) Los Chamelos, d) Punta Freddy, e) El Cernicalo, and f) Sal si puedes) are also represented. A) Depicting the sampling methods, and B) Bahía Cumberland and fisherman's boats. Photograph by Eduardo Sorensen and Alejandro Pérez-Matus.



neighboring South America, which has intrigued us as well as several other researchers (see Burridge et al., 2007). Formed by volcanic eruptions approximately 3.1 millions years ago, the Juan Fernández Archipelago resembles the Galapagos Islands with respect to the Equator coast.

As opposed to the terrestrial environment, which is facing several anthropogenic impacts through the introduction of invasive species, the marine realm, in particular the subtidal habitats of these Islands have been poorly studied. Moreover, few studies have described ecological interactions and their impacts that may drive species to co-exist in such isolated oceanic islands. However, important efforts have been made to determine geographical breaks, transitional areas, biogeographic patterns and most importantly, the fishery and biology of the Juan Fernández Lobster (*Jasus frontalis*) (reviewed by Arana, 2001). Taxonomic uniqueness and difficult access to these islands further limit insight into the ecology and biology of species that inhabit the islands. Our research trip represents a snapshot of the spatial distribution of marine subtidal species throughout Robinson Crusoe and Santa Clara Islands. These results and previous studies mayprovide the basis for models of future management and conservation strategies for subtidal habitats.

MAJOR DESCRIPTIONS:
OCEANOGRAPHIC AND MARINE FLORA AND FAUNA

One of the most prominent features of water circulation off continental Chile is the 300 to 400 Km of water mass called the Humboldt current, generated by the northward deflection of the West Wind Drift when it reaches the continent at approximately 45 º S. North of 40 º S and at 1000 km off the coast, two other water masses take over: the Subantarctic and the Subtropical currents. The latter warm mass (23-27 º C winter-summer temperature variation with salinities above the 34.5 0/00) flows southward over the northward cold water mass (10 to 18º C temperature variation with salinities of 32 to 34 0/00). This generates complex circulation and countercurrent regimes along the coastline, which are combined with strong southerly winds that generate a system of seasonally upwelled waters (Farina et al., 2005). In addition, oceanographic conditions off the northern coast of Chile are modified significantly during El Niño events (Camus, 1990; Thiel et al., 2007). Juan Fernández Archipelago, in turn, has surface temperatures above the 15º C with salinities of 34.3 0/00, and Subtropical water masses dominate over the Subantarctic ones in this region (Moraga & Argandoña, 2001).

As the Islands are characterized by steep slope, the rocky intertidal zone is considerably reduced (Figure 1). Therefore we concentrated our efforts in the sampling of the subtidal portions of the island by means of observational and photography-based data collection from several different sites throughout the archipelago. Our interest of study was principally related to community ecology describing abundance (biomass), size, density, and habitat type for key species of fish, shellfish, and algae from 2 to 35 m depth.

The subtidal habitat of the archipelago is characterized by rigid rock and volcanic boulder structures. Depending on the location of the site within the archipelago, topography is characterized by vertical walls, boulders or cobbles. Principally the "north-eastern" sites (Bahía Cumberland, El Frances, and Sal si Puedes; Figure 2 a, b, and c respectively) were highly eroded and were characterized by sand grains and small boulders. Big boulders, caves and vertical walls up to 40 m in depth principally characterize southern and more exposed sites (sites c and d, Figure 2).

Rocky reefs facilitate the settlement of some macroalgae and sessile invertebrate species. Different degrees of structural complexity found at the study sites form several microhabitats, which allow for differential organization of biotopes in small geographical areas. We found that benthic habitats are characterized by crustose algae that cover most of the deep portions of the volcanic rocks and bushy and erect brown macroalgae, predominantly the endemic *Padina fernandeziana*, *Dyctiota kunthii* and *Colpomenia sinuosa*, which form dense assemblages up to 20 m depth. Despite their abundance and dominance throughout the Chilean temperate coast, no kelp (Laminareales) was identified on the archipelago, possible due to the elevated seawater temperatures present in the island in respect to the continent. On the other hand, the introduced *Codium fragile* is abundant in almost all subtidal portions of protected bays we sighted. Another important component of the sea floor up to 35 meters depth is the cover of Vermetid gastropods throughout the archipelago; which can reach up 15 to 20 percent cover are characterize by mucus they generate for feeding.

All vertical walls of the islands are covered with colorful cnidarians and poriferans (Figure 3). Several zooanthids (*Parazoonthus juanfernandezi*) aggregate together and occur in vertical structures to trap their food. The sea cucumber (*Mertensiothuria platei*), a filter feeding macroinvertebrate is abundant at shallow depths and forms dense aggregations (4 individuals per m^2, approximately). Black corals (*Antipathes fernandezi*) are usually found deeper (Figure 4).

FIGURE 4. Black Coral. Photograph, Eduardo Sorensen.

FIGURE 5. Sea urchin and Vermetid grastropods. Photograph, Eduardo Sorensen

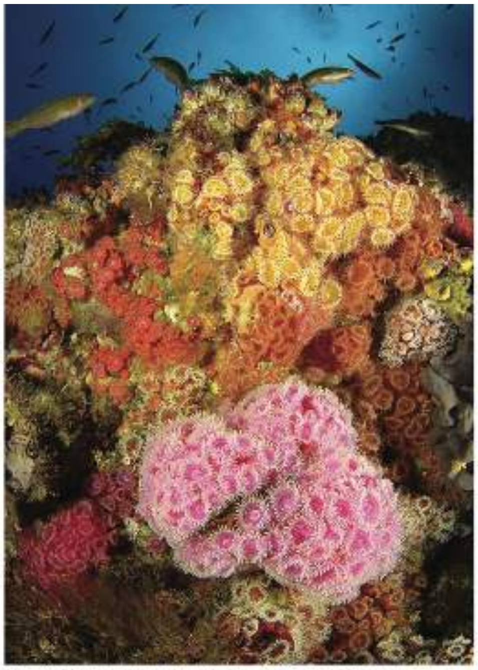


FIGURE 3. Diversity of cnidarians. Photograph by Eduardo Sorensen.

Mobile invertebrates are represented by the black sea urchin (*Centrostephanus rodgersii)*. High densities (around 5 individuals per m^2) of this echinoderm are predominantly found at sites protected from wave action. At these high densities urchins form "barrens", areas of substrate dominated by urchins and typically devoid of any macroalgae (Figure 5). We hypothesize that "urchin barrens", which alter environment and landscape, may generate profound impacts for a small island shelf as they may reduce potential habitat for the settlement of other benthic species such as lobsters.

Algae appear to change in abundance and coverage in the most exposed areas such as Santa Clara Island and Los Chamelos (sites c and d, Figure 2). Here the sea floor is steep with vertical walls and small seamounts fragmenting species composition. Shallow depths are typically characterized by turfing macroalgae such as the "ephemeral algae" *Enteromorpha intestinalis* and *Ulva* spp. Also present are: *Chaetomorpha* spp, and dense rhodophytes characterized principally by *Asparogopsis* sp., *Chantransia* spp, *Cryptonemia* sp., and *Ceramium rubrum*. Our observations at these sites represent similarities to the habitat characteristics found at some central to southern continental Chilean sites (personal observation, Pérez-Matus). At these exposed sites "loco" (*Concholepas concholepas*), the most important benthic resource of continental Chile, thrives with abundances of 3 to 4 individual per m^2, however there is no harvest permitted on the archipelago. The lobster (*Jasus frontalis)* was larger and more abundant in some patches and densely distributed at some sites characterized by caves and crevices such as in Chamelos and Punta Freddy (sites c and d, Figure 2). The starfish (*Astrotole platei*) is conspicuous and occurred in all study sites at low abundances.

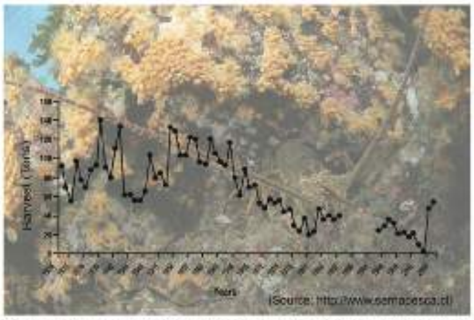
Similarly, as percent cover of sessile flora and fauna vary among sites, the distribution of fish fauna represents a mosaic in terms of assembly formation and size distribution. Depth seems to be the only major factor that clearly stratifies the vertical distribution of fishes, an observation that is intrinsic to other populations and fish communities. We identified more than 25 species of fish; the most abundant are represented by wrasses (*Malapterus reticulatus* and *Pseudolabrus gayi*), which are spread throughout the study sites at shallow depths forming dense schools. Pelagic fishes as "pampanito" (*Scorpis chilensis*) and mackerels such as the transpacific *Trachurus murphyi* and *Pseudocaranx chilensis* are highly abundant in the southern study sites such as El Frances, where dense schools were observed (Figure 6). Other pelagic such as the kingfish (*Seriola lalandi*) and *Mola mola* cruise around the islands and were found at most survey sites. Benthic-pelagic fishes such as the serranid *Caprodon longimanus* were abundant at intermediate depth (10 to 20 m), forming aggregations of 25 to 30 individuals (Figure 7) particularly in protected bays such as El Cernicalo (site e, Figure 2). The benthic-territorial blenny *Scarthichthys variolatus* and the colorful "cabrilla de Juan Fernandez" *Hypoplectrus semicinctum* (endemic) were also conspicuous, occupying almost all of the cave structures at shallow depths (2 – 15 m). Gobies, roughies, and moray eels occupy almost all of lobsters' refuges. Herbivorous girellids (*Girella albostriata*) were usually found in schools of 20 individuals (unusual for temperate herbivores) and graze over vast macroalgal gardens primarily at southern sites.

Mammals are also represented on the islands. Nearly driven to extinction in the 60's by fur-trade companies; the endemic sea lion (*Archtocephalus phillipi*) is now recovering. Recent censuses performed on the archipelago estimate an approximate population of 300 000 individuals and from March to November these pinnipeds migrate to search for food. Whales are also sighted during spring and summer.

The data compiled in this study combined with historical fisheries catch data will create a database describing historical resource use and current biological community structure. Interviews with fishermen, community members, and ecologists provide local knowledge of community structure changes through time. This data is employed to model ecological relationships describing trophic interactions and the flow of energy and biomass through a biological community using STELLA and Ecopath software. These models are being used to test a series of scenarios of differing management schemes altering larval dispersal rates, fish immigration and emigration rates, fishery catch estimates, and marine reserve protection. This allows for simulation of differing protection measures, seasonal closures, minimum capture size, and extraction levels to determine effective management and conservation of species.

MANAGEMENT & CONSERVATION:
APPLICATIONS FOR A MONOSPECIFIC FISHERY

The current lobster management plan used at the Juan Fernández Archipelago employs a seasonal closure (arbitrarily chosen) from May 15th until September 30th, a minimum cephalothorax harvest size of 11.5cm, and no capture of egg-carrying females. This minimum size limitation promotes the harvest of larger lobsters, thereby selecting for a smaller average size in the population. However, illegal harvest of undersized lobsters has been documented for consumption by fisherman and as bait for traps (Arana, 1987). The lobster fishery is considered to be mono-specific, however in the process several other species are targeted resulting in the harvest of nearly 300 pounds (136 kg) of fish a day for both bait and supply to the fresh



(Source: http://www.sernapesca.cl)

Figure 8. Total annual harvests of the Juan Fernandez's spiny lobster (*Jasus frontalis*). Including all the islands of the archipelago. Photograph by Eduardo Sorensen.

market (according to fishing community association of Juan Fernandez). The current lobster fishing effort is concentrated in the farthest areas of the archipelago in relation to the population centre and main port, Bahia Cumberland (site a, Figure 2). It is evident from harvest data (Figure 8) and anecdotal evidence from fishermen that despite current management regulations the lobster population has been severely depleted in most areas of the archipelago compared to pre-human inhabitation. For these reasons we presented information to the fishermen and community members of the Juan Fernández Archipelago about marine reserves (MRs) and marine protected areas (MPAs) and their potential benefits for the local lobster fishery and for species that are rare or face local extinction.

This presentation addressed the similarities of fish diversity between New Zealand and the Juan Fernández Archipelago irrespective of the long geographic distance that separate them. The dynamics of trophic interactions: how changing the abundance of one species may have dramatic effects on the abundances of other species with reference to the "loco"-mussels dynamic demonstrated at Las Cruces Marine Reserve in Chile (Castilla, 1999) were outlined. Benefits that MRs have demonstrated in many New Zealand locations for a closely related lobster species, *Jasus edwardsii* with regard to increased size and abundance inside MRs were presented. The conservation benefits of MRs for the selected and highly harvested lobster species are important as larger lobsters are targeted by the fishery. MRs as a potential source for larval export and emigrating juveniles and adults may also supplement adjacent populations, a process driven by density–dependence, commonly called the "spill-over" effect. Social and economic benefits of a MR for the archipelago could be dramatic, as the island currently is marketed as a dive destination and attracts tourism for its terrestrial national park. A marine reserve could provide an area of aquatic conservation that would attract naturalists, divers, conservationists and scientits helping to foster stewardship among the local community.

Figure 6. Abundance of Pelagic fish species.
Photograph by Alejandro Pérez-Matus.

In reviewing studies performed on the archipelago we suspect that the Juan Fernández Islands are a fragile ecosystem with extinction and speciation rates occurring over short temporal scales. A highly complex process may take place in such small islands. We hypothesize that the Juan Fernández Archipelago represents a mosaic of biotas depicted by open (relying on recruits from other populations) for endemic species and closed populations (self-recruiting) acting as sources from adjacent areas (even the farthest ones). Without understanding the complete dynamics of community structure and how populations are connected in such a small isolated area, management strategies are uncertain up to date however important for a sustainable fishery to persist.

Tyler D. Eddy, Fabian Ramirez*, and Alejandro Pérez-Matus**
School of Biological Sciences, Victoria University of Wellington
Wellington, New Zealand
*Facultad de Ecologia y Recursos Naturales, Universidad Andrés Bello
**E-mail: *alejandro.perezmatus@vuw.ac.nz*

Also Published in *JMBA*

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Figure 7. School of cabrilla (*Caprodon longimanus*).
Photograph by Alejandro Pérez-Matus.

Appendix E

Eddy TD, Gardner JPA (2009) Trophic modeling of a temperate marine ecosystem throughout marine reserve protection in New Zealand. Ecopath 25 Years Conference Proceedings: Extended Abstracts.

TROPHIC MODELING OF A TEMPERATE MARINE ECOSYSTEM THROUGHOUT MARINE RESERVE PROTECTION IN NEW ZEALAND[1]

TYLER D. EDDY
JONATHAN P.A. GARDNER
Centre for Marine Environmental and Economic Research. School of Biological Sciences, Victoria University of Wellington, Coastal Ecology Laboratory, 396 The Esplanade, Island Bay, New Zealand;
tyler.eddy@vuw.ac.nz; jonathan.gardner@vuw.ac.nz

Marine Reserves (MRs) in New Zealand are being monitored and investigated to determine implications for conservation and management strategies. This research project employs a variety of techniques to answer questions about how MRs impact biological communities and what this means for the management of coastal resources. Underwater research at three central New Zealand MRs (Kapiti MR, Long Island MR and the newly implemented Taputeranga MR; see Figure 1) conducting seasonal size and abundance surveys of reef fish, invertebrates and algae at both protected and unprotected locations is used to determine biomasses of trophic groups. Monitoring data also exists prior to and throughout MR protection, which is used to determine ecosystem response to protection in temperate central New Zealand waters. This monitoring information is used in combination with biological data from the literature to describe trophic linkages within the ecosystem.

Figure 1. Map of New Zealand showing marine reserve locations with three study sites shown in red (Kupe/Kevin Smith is now known as Te Taputeranga). Image modified with permission from New Zealand Department of Conservation.

An ecosystem model that was created for Te Tapuwae o Rongokako MR located midway up the east coast on the North Island, New Zealand has identified that the region is relatively poor in invertebrate biomass when compared to Leigh MR, which lies further north (Pinkerton *et al.*, 2008, Lundquist & Pinkerton 2007). It was determined that the diet of lobsters is composed of a large amount of macroalgae, which has not been observed in other regions of New Zealand. This ecosystem appears to be strongly influenced by lobster abundance, which has been increasing since implementation of the MR.

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This study is focused in the Cook Strait region located between the North and South Islands, which is characterized by colder waters in comparison to the north and is a highly dynamic area where three ocean currents converge. Temporal data is also used to evaluate ecosystem response to MR protection. This approach allows for an understanding of how MR design and placement, fisheries regulations and coastal resource use affect the dynamics of a biological community. Each of three marine reserves investigated have different designs with respect to boundaries, size and shape. Commercial, recreational and traditional fisheries for reef fish and invertebrate species are important in these regions and we wish to understand how ecosystems respond to MR protection and the impact of factors such as size, placement and design.

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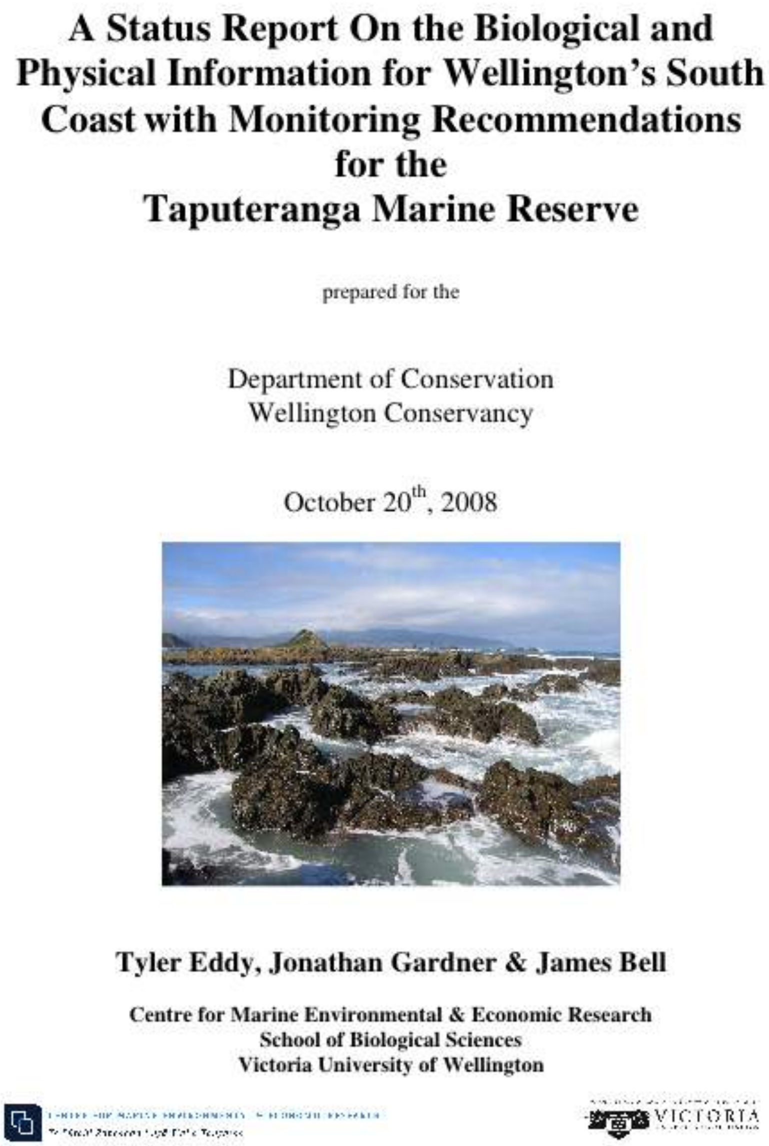
Appendix F

Eddy TD, Gardner JPA, Bell JJ (2008) A Status Report on the Biological and Physical Information for Wellington's South Coast with Monitoring Recommendations for the Taputeranga Marine Reserve. New Zealand Department of Conservation Report. (Front page only; available to download online at: http://www.doc.govt.nz/publications/conservation/marine-and-coastal/marine-protected-areas/status-report-on-wellingtons-south-coast/**)**



A Status Report On the Biological and Physical Information for Wellington's South Coast with Monitoring Recommendations for the Taputeranga Marine Reserve

prepared for the

Department of Conservation
Wellington Conservancy

October 20th, 2008

Tyler Eddy, Jonathan Gardner & James Bell

Centre for Marine Environmental & Economic Research
School of Biological Sciences
Victoria University of Wellington

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