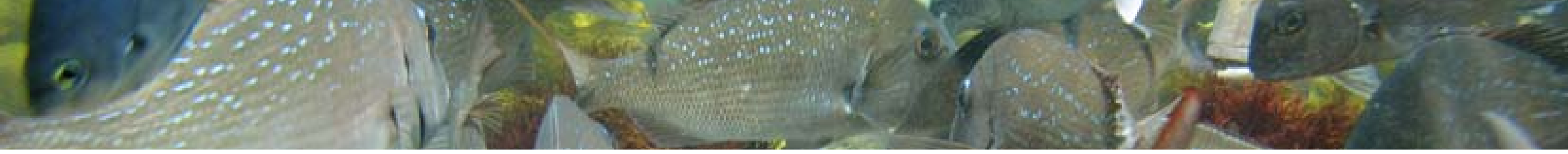

Poor Knights Islands Marine Reserve and Mimiwhangata Marine Park fish monitoring 2009

Department of Conservation
Te Papa Atawhai

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Review process

This document has been submitted to one internal and one external expert reviewer. Reviewers were selected because of their expertise in marine science and marine conservation. This review process was undertaken to ensure that methodologies and recommendations within this report are robust and in line with best practice. This report has been reviewed by:
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Dr Nick Shears

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Cover photo: Baited video monitoring equipment in action at the Poor Knights Islands Marine Reserve
Photo: Paul Roux De Buisson

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CONTENTS

FIGURES

TABLES

Executive summary

In New Zealand and internationally there is currently strong interest in the use of marine reserves and fisheries regulations to protect biodiversity values and marine ecosystems. This study uses baited video to measure the relative abundance of reef fish at the Poor Knights Marine Reserve (full no take protection), the Mimiwhangata Marine Park (partial protection), Mimiwhangata, North Cape, Cape Karikari and Cape Brett (open to fishing). Using existing data and recent survey data, the performance of these three alternative management strategies is evaluated.

At the Poor Knights from 1981 to 1998 all commercial fishing nets and longlines were prohibited and recreational fishers were allowed partial access. Baited video data indicates that snapper populations did not recover under partial protection and were not significantly greater than at Cape Brett and the Mokohinau Islands which were open to recreational and commercial fishing.

This study found that in 2009 after more than ten years of no take protection snapper counts were 14 times greater than in 1998 before the marine reserve became fully no-take. In 2009 the average snapper length in baited video samples at the Poor Knights Islands Marine Reserve was 373 mm ± 16, significantly greater than all other locations. In 2009, after more than ten years of protection estimated snapper biomass per baited video camera was 528% greater than when the reserve was first sampled in summer 1999.

Snapper populations at North Cape, Cape Karikari and Cape Brett were significantly less abundant and fish were smaller in size than at the Poor Knights. In areas that were open to fishing large mature snapper were relatively rare. The Poor Knights Islands Marine Reserve has resulted in a significant increase in the size and abundance of snapper. A longer time series of data, collected using a range of methods, is required to resolve the effects of protection on other species of reef fish, algae and invertebrate communities.

Snapper abundance and size within the Mimiwhangata Marine Park were compared with fished sites at Mimiwhangata. These comparisons indicate that snapper abundance within the marine park was not significantly different than adjacent areas of coast open to fishing. This study supports previous investigations and concludes that the Mimiwhangata Marine Park fishing restrictions are not achieving their goal of protecting biodiversity while allowing for limited recreational take.

Marine reserves have consistently demonstrated a capability to protect reef fish assemblages and algal and invertebrate communities both within New Zealand and internationally. Until recently little was known about the capability of partial protection mechanisms to achieve biodiversity objectives while allowing for limited harvesting. Baited video monitoring at the Poor Knights Islands Marine Reserve and at Mimiwhangata Marine Park have again shown that partial protection mechanisms do not effectively protect some species of reef fish.

At the Poor Knights Islands Marine Reserve it is currently unknown whether the snapper population is continuing to increase in abundance and size after more than 11 years of no take protection. Further sampling with alternative methodologies is required to resolve whether the snapper population has reached a plateau or is still growing. Little is known about the effects this large increase in predatory biomass will have on other components of the ecosystem at the Poor Knights. Further investigations into habitat utilisation and the diet of snapper are necessary to better understand any potential effects on benthic species and habitats at the Poor Knights Islands Marine Reserve.

Introduction

A marine protected area (MPA) is a zone that is managed so that marine habitats and ecosystems are healthy and functioning properly. Marine reserves are one of the highest forms of MPA protection and the New Zealand government has committed to protecting representative, rare, outstanding and biologically important ecosystems with marine reserves. In New Zealand and internationally there is also strong interest in the use of partially protected areas to protect biodiversity while minimising the impacts on existing users. In New Zealand an MPA must meet a minimum protection standard and “adequately protect” marine biodiversity (Ministry of Fisheries & Department of Conservation 2008). Located in Northland, New Zealand, the Poor Knights Islands Marine Reserve has now been completely no take since October 1998. The Mimiwhangata Marine Park has been partially protected with fishing restrictions since 1984. This report evaluates marine reserves and partial fisheries restrictions as management tools to adequately protect biodiversity. Relative reef fish abundances are compared between the Poor Knights Islands Marine Reserve (no take protection since 1998), Mimiwhangata Marine Park (partial protection) and fished reference locations at Mimiwhangata, North Cape, Cape Karikari and Cape Brett (open to fishing). Knowledge about the capability of these different management mechanisms to adequately protect biodiversity and to meet the protection standard is crucial to enable the design of an effective network of marine protected areas in the future.

POOR KNIGHTS ISLANDS MARINE RESERVE

The Poor Knights Islands are located in Northland New Zealand, 24 km offshore (figure 3). In 1981 the Poor Knights Islands Marine Reserve was established under the Marine Reserves Act with the intent of protecting the Poor Knights Islands’ unique biodiversity. The Poor Knights Islands Marine Reserve extended 800 m out from the islands, and from 1981 to 1998 all commercial fishing nets and long lines were prohibited. However, recreational fishers were permitted to use unweighted single hook lines and to troll and spear within a large portion of the Poor Knights Islands Marine Reserve. In May 1997, the Minister of Conservation announced plans to change the fishing regulations at the Poor Knights to take effect on 1 October 1997 (partial closure), which would implement a complete fishing exclusion extending to 800 m from the islands, coming into effect from 1 October 1998.

Monitoring of reef fish was initiated in 1998 (Denny & Babcock 2003) by the Department of Conservation (DOC) and the University of Auckland to measure the responses arising from implementing total no take marine reserve protection. Baited videos (Willis and Babcock 2000) have been used to survey reef fish at the Poor Knights from the spring of 1998 to the autumn of 2009 (Denny et al 2003, Denny & Shears 2004). Results have indicated that following the implementation of full no take marine reserve status at the Poor Knights in 1998, snapper (*Pagrus auratus*), the most heavily targeted recreational and commercial fish species throughout north-eastern New Zealand, showed significant increases in size and abundance. A biological survey and monitoring strategy for the Poor Knights Islands Marine Reserve was developed in 2008 (Roux De Buisson 2008) to guide future survey and monitoring work.

This investigation builds on previously published data and the Poor Knights Islands Marine Reserve monitoring program by using baited underwater video equipment to investigate relative reef fish abundance at the Poor Knights Islands Marine Reserve after more than ten years of protection. Comparisons of relative reef fish size and abundance are made between the Poor Knights, and four fished reference sites; North Cape, Cape Karikari Cape Brett and Mimiwhangata.

MIMIWHANGATA MARINE PARK

The Mimiwhangata Marine Park is situated on the east coast of Northland, south of Oakura and north of Tutukaka (figure 3). The marine park extends 1km offshore between Paparahi Point and Te Ruatahi Island and extends beyond Rimariki Island. The Mimiwhangata Marine Park was established in 1984 by Lion Breweries and the Mimiwhangata Charitable Trust. The vision was to manage the marine area in an environmentally sustainable way and to protect the marine ecosystem for the enjoyment of all New Zealanders (Grace 1984).

Special rules have existed in this area since 1984 and were designed to protect reef dwelling fish and other species that are vulnerable to overfishing, are long lived or have low reproductive rates (such as paua and rock oysters). All commercial fishing is prohibited. Nets and long lines are not permitted, including those set by kontiki and kites. However a special clause allowed commercial crayfish potting and long lining until October 1993. Amateur fishers may use only the following methods: un-weighted, single hooked lines, trolling, spearing and hand picking. Only those species of fish and shellfish listed in the "Permitted Species List" may be taken. These species are: barracuda, billfish (all types), blue maomao, flounder (all types), grey mullet, yellow eyed mullet, gurnard, kahawai, kingfish, mackerel (all types), piper (garfish), shark (all types), snapper, sole, trevally. Potting for rock lobsters is permitted providing that only one pot per person, party, or boat is used. For permitted species normal regulations apply regarding daily bag limits, sizes, closed seasons, condition, shelling and pot escape gaps.

Investigations of reef fish abundance at Mimiwhangata were carried out in 2002 and 2003 (Denny & Babcock 2002, Usmar et al. 2003). Both investigations found that snapper (*Pagrus auratus*) abundances were not significantly different between the marine park and reference locations outside of the marine park. Both reports concluded that despite the exclusion of commercial fishers and restrictions on recreational fishing, the partial fishing regulations at Mimiwhangata have been ineffective as management tools to protect the marine ecosystem within the park. Both reports consistently found that reef fish abundance at Mimiwhangata reflected that of typically fished coasts elsewhere in northland. Crayfish (*Jasus edwardsis*) populations have also been monitored at Mimiwhangata since 1977. Long term trends indicate that crayfish have not recovered since the marine park was established. Comparisons of crayfish counts between no take marine reserves and the Mimiwhangata Marine Park have clearly demonstrated that preventing commercial cray fishing while allowing recreational cray fishing at Mimiwhangata has had little benefit to populations of crayfish (Shears et al. 2006).

This investigation will reevaluate the effectiveness of the Mimiwhangata Marine Park fishing regulations to protect carnivorous fish while allowing for limited recreational fishing. Reef fish monitoring at Mimiwhangata in this investigation has been undertaken during the same season and with the same methodology as monitoring at North Cape, Cape Brett, Cape Karikari and the Poor Knights Islands Marine Reserve. The effectiveness of alternative management strategies will be compared between offshore island, exposed mainland (capes) and coastal locations.

Methods

BAITED VIDEO

A baited underwater video system has been used to monitor reef fish populations in New Zealand for more than ten years (Willis & Babcock 2000). This baited video system has been used to assess the abundance and size distribution of the snapper *Pagrus auratus* and blue cod *Parapercis colias* inside and outside marine reserves in New Zealand (Davidson 2001, Taylor et al. 2003, Willis et al. 2003, Denny & Shears 2004, Davidson & Richards 2005a, Davidson & Richards 2005b, Sivaguru 2007). Over the years, changes have occurred in the design and construction of the baited video system. The majority of these changes have been restricted to changes in materials used for the tripod and small refinements in design (Denny et al. 2003, Denny & Shears 2004, Langlois et al. 2008). The evolution of the baited video system used in the present study is described in (Langlois et al. 2008). The baited video system used in 2009 sits on the substrate and is held upright by a pressure buoy. Previous baited video sampling methods and analyses at the Poor Knights (Willis & Babcock 2000, Denny & Babcock 2003, Denny & Shears 2004) and at Mimiwhangata (Denny & Babcock 2002, Usmar et al. 2003) were repeated.

Baited underwater video sampling involves dropping a camera attached to a frame (fig. 1) into the water and filming fish as they are attracted to a bait pot (Willis & Babcock 2000). Baited videos were submerged around thirty times at a monitoring location for a thirty minute sampling period. Bait pots were filled with fresh bait (pilchards) before each deployment and drops were placed a minimum distance of 50 m from another deployment location to avoid potential effects on fish behavior due to interference between baited videos. The footage was played back and the maximum number of each fish species recorded in a frame over the 30 minute film sequence was recorded. The frame containing the maximum number of snapper was analysed further. Still frames were saved from the video sequence and calibrated against a scale bar of known length and a bait container of known length within the baited video's field of view (fig. 1). Because the camera is not bi-focal, care was taken to accurately measure fish length. Fish were measured using three point calibration and were only measured when they were at the same level as a calibration point of known length, usually the bait container. Snapper length from the video frame with the maximum number of snapper was converted to wet weight biomass using $W = aL^b$ where W is weight(g), L is length , a is 7.194×10^{-5} and b is 2.793 (Taylor & Willis 1998).

Drop sites were fixed GPS waypoints typically on sand immediately adjacent to rocky reefs. A research vessel approached a fixed historical waypoint, the bottom was confirmed to be sand immediately adjacent to rocky reef with a Humminbird, transducer mounted side scan sonar (fig. 2) and then the baited video was dropped. In situations where the sampling site was over rocky reef substrate an additional quick release

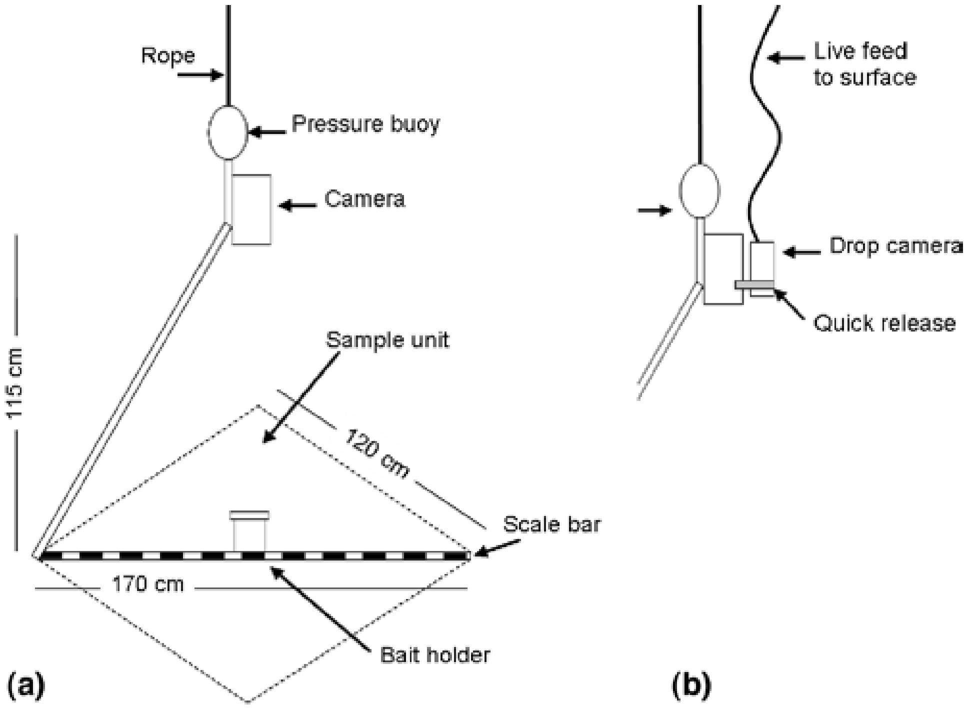


Figure 1: Remote baited video system used in 2009. (a) This system sits on the substrate and is kept upright by a pressure buoy. The camera films fish attracted to the bait. (b) A quick release camera supplying real time footage to the surface is attached allowing the baited video to be placed in complex habitats such as kelp forests and patch reef systems. The umbilical camera is removed once the baited video unit is in place. The above diagram has been taken from Langlois et. al (2008).

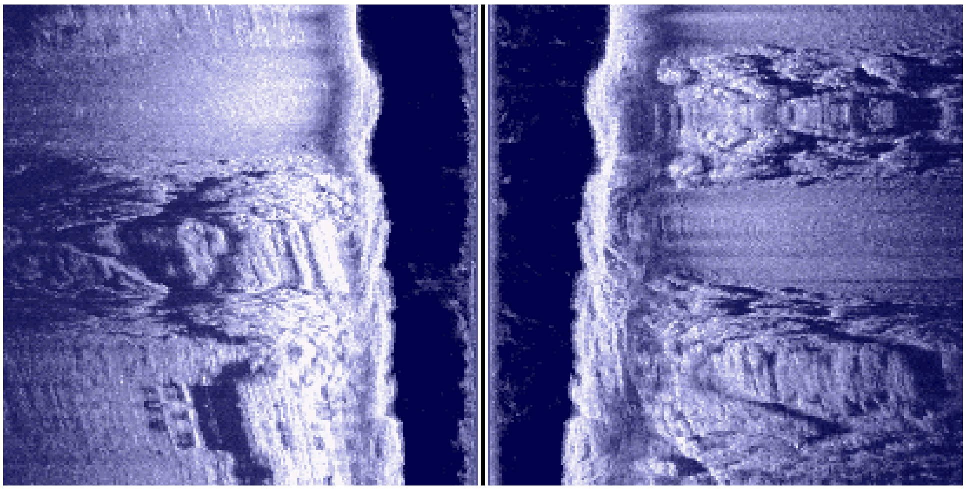


Figure 2: Humminbird sidescan sonar image used to determine the location of drop sites on sand immediately adjacent to rocky reefs. Smooth areas are sand and rough areas are reef.

camera with a live video feed to the research vessel was attached to the baited video (fig. 1). This enabled the operators to check that the baited video camera field of view was not obscured by seaweed.

Patterns in reef fish abundance may vary according to fluctuations in recruitment, change in habitats, physical forces (e.g., sea condition, water temperature), food resources, ontogenetic changes in habitat requirements, time of day, reproductive behavior, variation in mortality rates through predation, inter-specific interactions and the degree of fidelity fishes have to reefs (Kingsford and Battershill 1998). All of these factors vary through space and time and may affect baited video counts. The response of carnivorous species to the baited video apparatus at any point in time may also be affected by tidal cycles, depth and the position of the sampling station in relation to the reef structure and current patterns. In order to avoid bias, sampling locations were chosen haphazardly and represent a range of depths and positions in relation to reef structure and current flow. Sampling was undertaken throughout all tidal cycles and sand habitat adjacent to reef has been targeted at all locations where possible. By sampling using consistent methods over long time scales we can detect the relative influences of protection and fishing on reef fish populations, particularly snapper.

STATISTICAL ANALYSIS

The baited video methodology is particularly well suited to measuring the relative abundance of large mobile demersal predators such as snapper. However the baited video data are counts and therefore do not satisfy the assumptions of normality and homogeneity of variance that are required by ANOVA. Therefore, the baited video data was analysed using the Poisson distribution using the GENMOD procedure in SAS to obtain unbiased estimates of relative abundance for dominant carnivorous species. See (Willlis et al. 2000) for a more detailed description of this analysis. Size data are continuous variables and satisfy assumptions of normality and homogeneity of variability required by ANOVA. Size data were analysed using a one way ANOVA procedure in sigmaplot.

MONITORING SITES

The original Poor Knights monitoring program was set up in 1998 and used baited video at the Poor Knights, Cape Brett and the Mokohinau Islands (Denny et al. 2003). These locations were considered to be comparable rocky reef environments influenced by the East Auckland Current (Stanton 1973) comprising reef fish assemblages with a distinctive subtropical influence. The rationale behind including Cape Brett and the Mokohinau Islands in the monitoring program was that any difference in reef fish abundances over time at the Poor Knights Islands Marine Reserve that did not occur at the fished reference locations could be more strongly attributed to the cessation of fishing rather than regional wide fluctuations in reef fish populations. Monitoring carried out in 2009 expands on the original program by including baited video sites at Cape Karikari and North Cape (figs. 3–7) which are open to fishing. The total number of baited video drops at each of the sampling locations from 1998 through to 2009 is shown in table 1. The Mokohinau Island sites were not sampled in 2009 due to logistical difficulties.

Mimiwhangata was divided into eight areas (fig. 4)—four areas inside the marine park and four areas outside the marine park (two at either end) (Denny & Babcock 2002, Usmar et al. 2003). Reef fish populations inside the marine park areas were compared with reef fish populations outside the marine park. The total number of drops at Mimiwhangata is given in table 1. This investigation followed the same sampling design as previous investigations allowing for a direct comparison of results.

TABLE 1: NUMBER OF BAITED VIDEO DROPS AT EACH MONITORING LOCATION.

DATE	POOR KNIGHTS	CAPE BRETT	MOKOHINAU	CAPE KARIKARI	NORTH CAPE	MIMIWHANGATA
1998 Winter	30					
1999 Summer	31					
1999 Winter	29	35	31			
2000 Summer	30	31	33			
2000 Winter	30	30	30			
2001 Summer	30	30	30			
2001 Winter	32	30	30			
2002 Summer	32	32	29			
2002 Winter	31	31	30			30
2003 Summer						30
2004 Summer	31		29			
2009 Summer	28	25		23	26	29

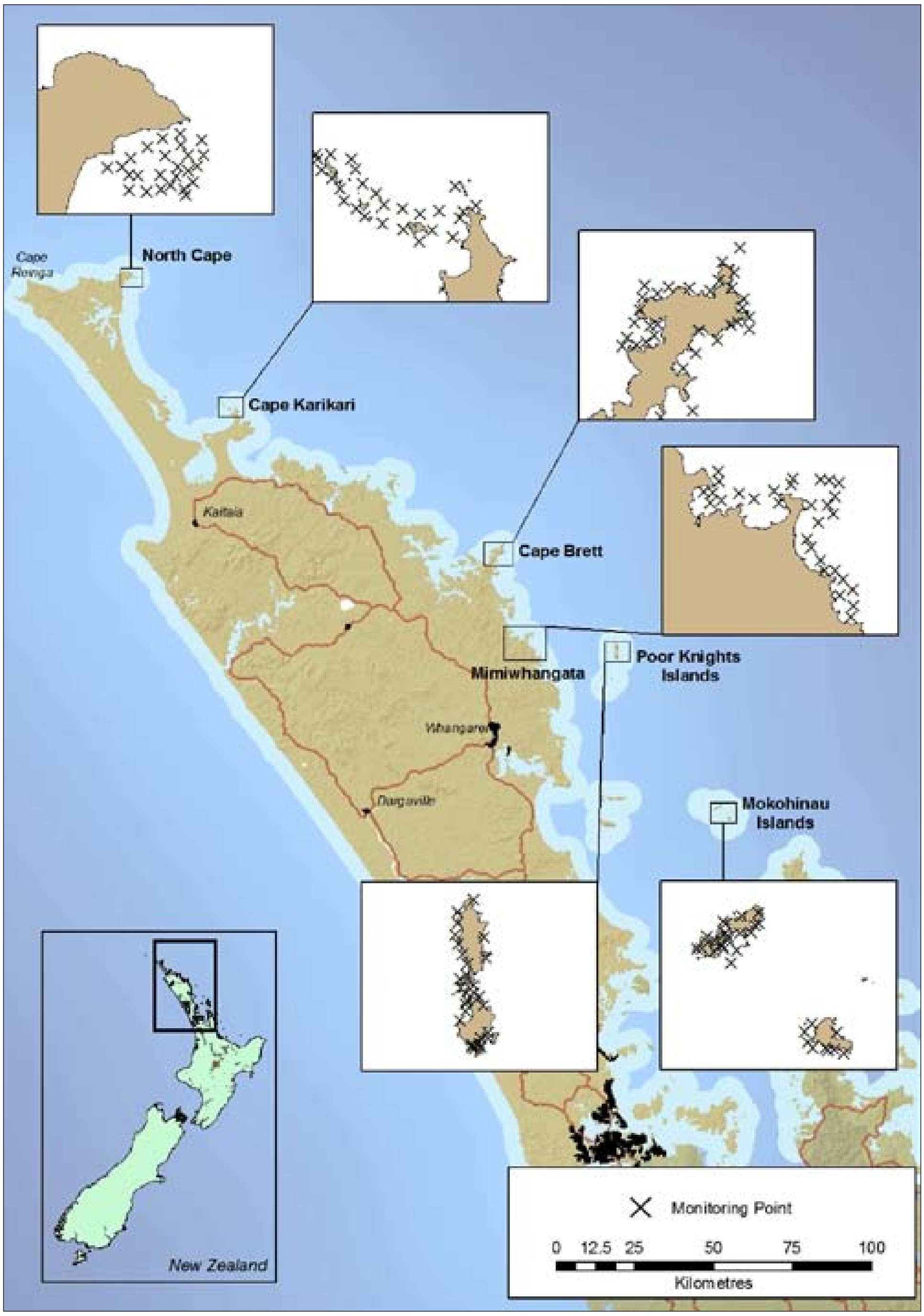


Figure 3: Location of baited video monitoring sites in Northland. The Poor Knights Islands Marine Reserve (no take), Mimiwhangata Marine Park (partial protection) and fished reference locations at Mimiwhangata, North Cape, Cape Karikari and Cape Brett (open to fishing). The Mokohinau Islands were not sampled in 2009

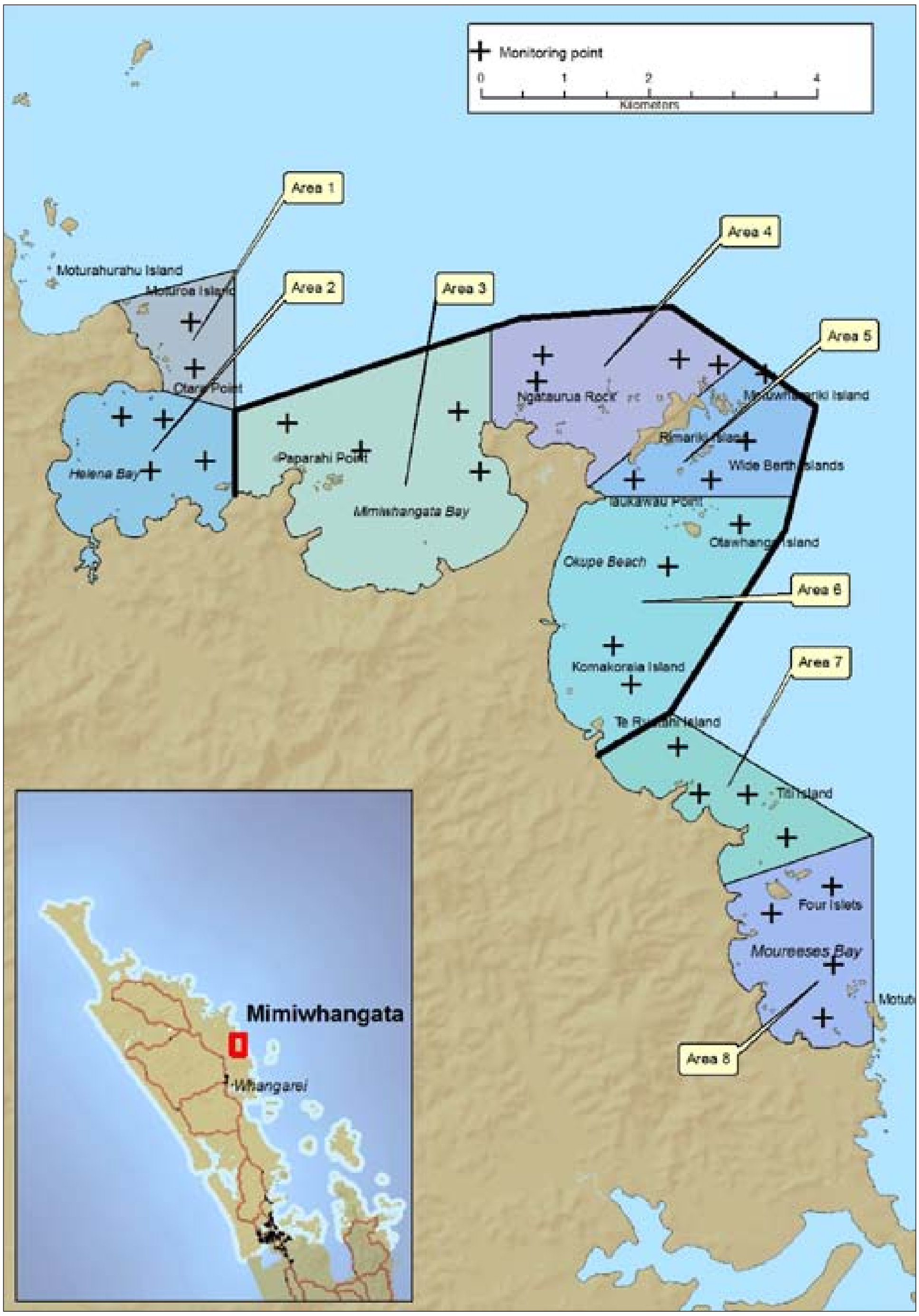


Figure 4: Baited video monitoring points at Mimiwhangata in 2009. The marine park boundary is marked with the bold line. Areas 3–6 are inside areas 1,2,7,8 are outside the marine park.

Results

POOR KNIGHTS ISLANDS MARINE RESERVE

Snapper counts for all locations and years are shown in figure 5. Data from 1998 to 2004 have been reproduced from (Denny & Shears 2004). Bi-annual sampling of snapper has shown strong seasonal trends. Summer recorded counts were consistently significantly higher than winter counts. To resolve the effects of no take protection on reef fish through time, summer and winter counts have been analysed and graphed separately to remove the influence of seasonality among years. The average maximum snapper counts for winter and summer surveys are presented separately in figures 6 and 7.

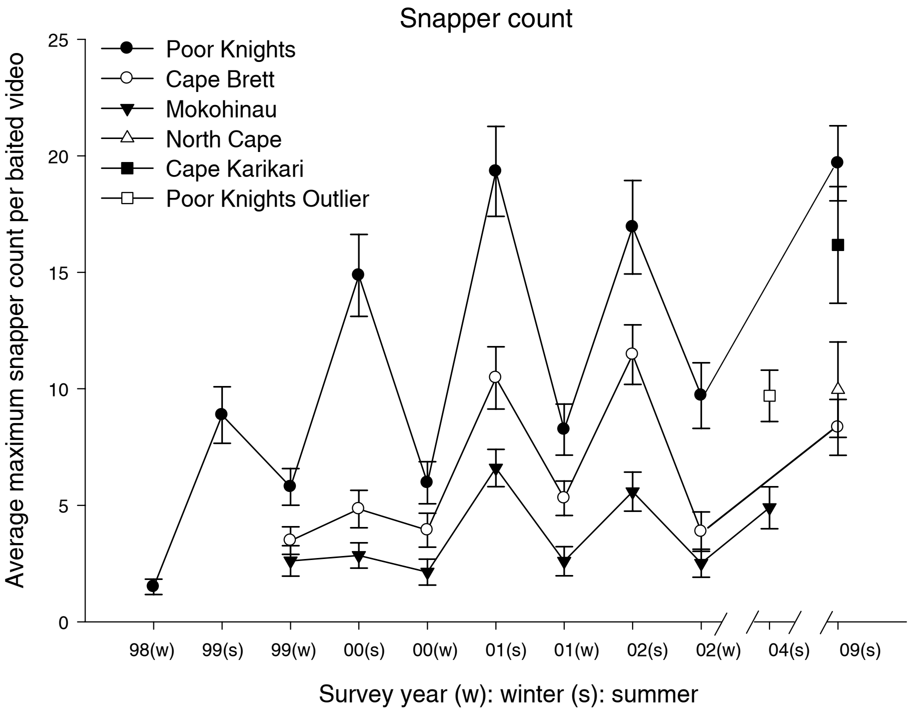


Figure 5: Average maximum number of snapper per baited video from all locations. Winter (September–December) and summer samples (February–June) with standard error bars.

Prior to implementation of no-take status in October 1998, the average maximum snapper (ams) count from winter baited video samples in 1998 was 1.5 ams. In winter of the next year (1999), after the marine reserve had been no take for 11 months, the average maximum snapper count increased significantly to 5.79 (ams). Over the next three years of winter sampling at the Poor Knights snapper counts increased steadily reaching 9.71 (ams) by the winter of 2002 significantly greater than the Mokohinau Islands and Cape Brett. Winter snapper counts at the fished reference locations of Cape Brett and the Mokohinau Islands over the same period did not change significantly.

At the Poor Knights average maximum snapper counts more than doubled from 8.87 (ams) in the summer of 1999 to 19.3 (ams) in the summer of 2001 after three years of protection. The summer sample in 2004 is treated as an outlier (PK Outlier in fig. 5) as the timing of sampling was up to three months later than other summer samples and the water temperature was significantly lower than previous summer samples. At the Poor Knights average maximum snapper abundance in 2009 was 19.7(ams) ± 1.6 the highest recorded abundance since monitoring began, 14 times higher (upper CL 27.5, lower CL 7.18) than in 1998. Snapper abundances were statistically compared between the Poor Knights and all other locations monitored in 2009 using ratios (see methods section) and are presented in table 2. For example a ratio of 0.42 at Cape Brett indicates that the average number of snapper per baited video at Cape Brett was 42% that of the Poor Knights. A p-value of less than 0.05 indicates a statistically significant difference. Average snapper abundance at the Poor Knights was significantly greater than all other locations in 2009 except Cape Karikari where average snapper abundances were 82% that of the Poor Knights. However average snapper size at Cape Karikari was 196 mm ± 11 much smaller than the average size of snapper at the Poor Knights which was 373 mm ± 16 (table 5).

TABLE 2: A STATISTICAL COMPARISON OF AVERAGE SNAPPER ABUNDANCE AT THE POOR KNIGHTS WITH ALL OTHER LOCATIONS SAMPLED IN 2009. ALL LOCATIONS HAVE BEEN COMPARED WITH THE POOR KNIGHTS USING RATIOS. A RATIO OF 0.5 AT NORTH CAPE FOR EXAMPLE INDICATES THAT AVERAGE SNAPPER ABUNDANCE PER BAITED CAMERA AT NORTH CAPE WAS 50% THAT OF THE POOR KNIGHTS. P-VALUES OF LESS THAN 0.05 INDICATE A STATISTICALLY SIGNIFICANT DIFFERENCE.

LOCATION	RATIO	UPPER CL	LOWER CL	P - VALUE
Cape Brett	0.424094	0.623342	0.288535	<.0001
Cape Karikari	0.821848	1.13698	0.594059	**0.2362**
Mimiwhangata (MP)	0.320652	0.4828	0.212961	<.0001
North Cape	0.506161	0.727527	0.352151	0.0002
Poor Knights	1	1	1	

TABLE 3: A STATISTICAL COMPARISON OF SNAPPER ABUNDANCES BETWEEN THE POOR KNIGHTS IN 2009 AND ALL PREVIOUS BAITED VIDEO SAMPLES TAKEN AT THE POOR KNIGHTS. (W) WINTER SAMPLE, (S) SUMMER SAMPLE. NON SIGNIFICANT P-VALUES ARE IN BOLD.

SURVEY	RATIO	UPPER CL	LOWER CL	P -
1998(w)	0.071140	0.139260	0.036342	<.0001
1999(s)	0.452395	0.619395	0.330421	<.0001
1999(w)	0.294375	0.427620	0.202649	<.0001
2000(s)	0.755406	0.992750	0.574805	0.0442
2000(w)	0.303189	0.436641	0.210524	<.0001
2001(s)	0.982357	1.269181	0.760353	**0.8918**
2001(w)	0.419203	0.576657	0.304741	<.0001
2002(s)	0.860536	1.118567	0.662028	**0.2616**
2002(w)	0.493368	0.669824	0.363397	<.0001
2004(s)	0.492875	0.671388	0.361826	<.0001
2009(s)	1	1	1	

Average maximum snapper abundance at the Poor Knights in 2009 was statistically compared with all previous baited video samples (table 3). Poor Knights snapper counts in 2009 were significantly greater than all previous samples except summer samples taken in 2001 and 2002

TABLE 4: AVERAGE MAXIMUM SNAPPER COUNTS PER BAITED VIDEO AT ALL LOCATIONS AND STANDARD ERROR. (W): WINTER, (S):SUMMER. MP (MARINE PARK), OMP (OUTSIDE MARINE PARK)

DATE	POOR KNIGHTS	CAPE BRETT	MOKOHINAU	CAPE KARIKARI	NORTH CAPE	MIMI (MP)	MIMI **(OMP)**
1998 (w)	1.5(0.3)						
1999 (s)	8.9(1.2)						
1999 (w)	5.8(0.8)	3.5(0.6)	2.6(0.65)				
2000 (s)	14.9(1.8)	4.8(0.8)	2.8(0.54)				
2000 (w)	6(4.9)	3.9(0.7)	2.1(0.56)				
2001 (s)	19.3(10.6)	10.5(1.3)	6.6(0.8)				
2001 (w)	8.3(1.1)	5.3(0.7)	2.6(0.6)				
2002 (s)	17(2)	11.5(1.3)	5.6(0.8)				
2002 (w)	9.7(1.4)	3.9(0.9)	2.5(0.6)			4.4(1)	4.6(1)
2003 (s)						3.9(0.8)	4.6(1)
2004 (s)	9.7(1.1)		4.9(0.9)				
2009 (s)	19.7(1.6)	8.3(1.2)		16.2(2.5)	10(2)	6.9(1.3)	6.31(1)

TABLE 5: AVERAGE MAXIMUM SNAPPER FORK LENGTH (MM) AT ALL LOCATIONS AND STANDARD ERROR BARS

DATE	POOR KNIGHTS	CAPE BRETT	MOKO HINAU	CAPE KARIKARI	NORTH CAPE	MIMI(MP)	MIMI (OUTSIDE MP)
1998 (w)	296(8.5)						
1999 (s)	275(4.2)						
1999 (w)	298(6.6)	216(3.6)	237(4.7)				
2000 (s)	301(6.4)	202(3.7)	229(4.4)				
2000 (w)	310(5.2)	219(5.5)	216(6.1)				
2001 (s)	330(4.9)	200(3.7)	216(2.7)				
2001 (w)	337(5.2)	202(4.3)	238(4.2)				
2002 (s)	310(3.2)	222(2.4)	227(3.4)			204(3.7)	199(5.8)
2002 (w)	300(4.0)	205(3.5)	236(5.1)				
2003 (s)						233(4.2)	237(6.2)
2004 (s)	344(4.9)		253(4.6)				
2009 (s)	373(16)	229(11)		231(11)	196(11)	264(4.2)	290(7.7)

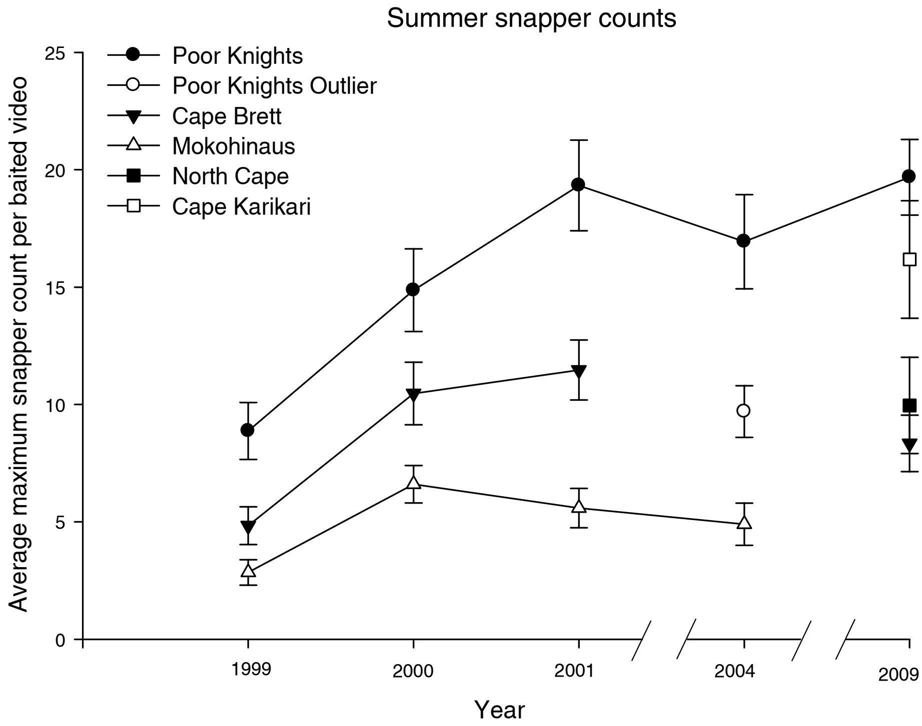


Figure 6: Average maximum snapper counts per baited video at all locations in summer and standard error bars.

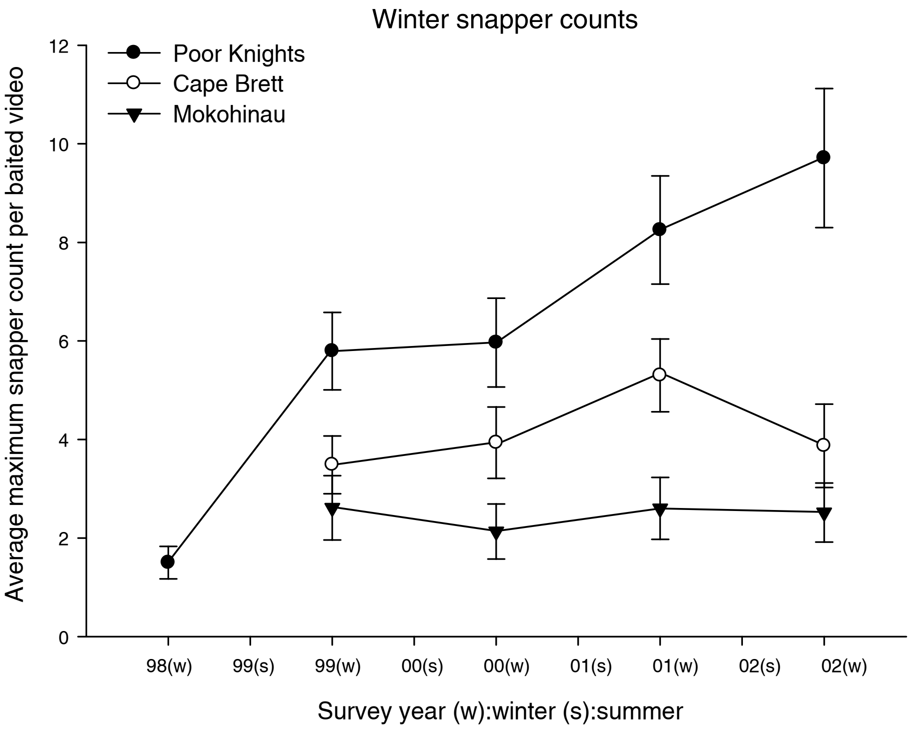


Figure 7: Average maximum snapper counts per baited video at all locations in winter and standard error bars.

Average snapper lengths are displayed in table 5 and fig. 8. Figure 10 compares the size frequency distributions of snapper populations at the Poor Knights, Cape Brett, Cape Karikari and North Cape in 2009. Figure 8 shows that at the Poor Knights in 1998 the mean length of snapper was significantly greater than fished reference locations prior to the reserve becoming no take. Since 1998 at the Poor Knights average snapper length has increased significantly from 293 mm in 1998 to 373 mm in 2009. At the Poor Knights the smallest fish recorded was 212 mm and the largest fish recorded was 752 mm. At the Poor Knights there were much higher abundances of fish over the minimum legal recreational size limit of 270 mm compared with all other fished locations. There was a distinct absence of small (<200 mm) snapper at the Poor Knights.

Over the same period average snapper length at fished reference locations has remained relatively unchanged. Average snapper lengths for all fished reference locations were below the minimum recreational size limit of 270 mm. At North Cape, Cape Karikari and Cape Brett recorded snapper were generally smaller than the minimum legal size and large mature fish were rare.

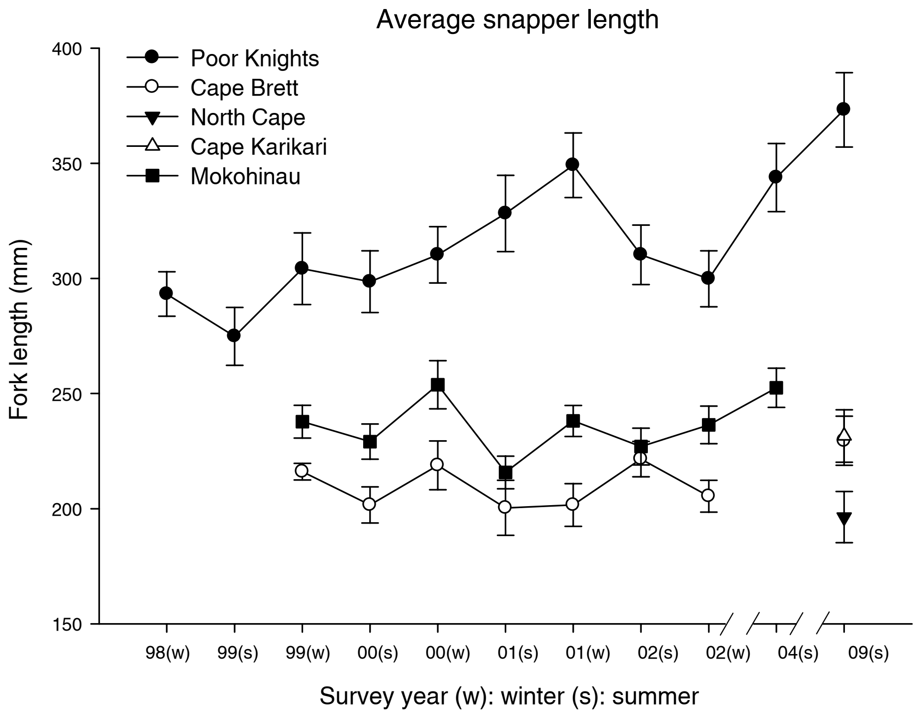


Figure 8: Average maximum snapper length (fork length) at all locations and standard error bars.

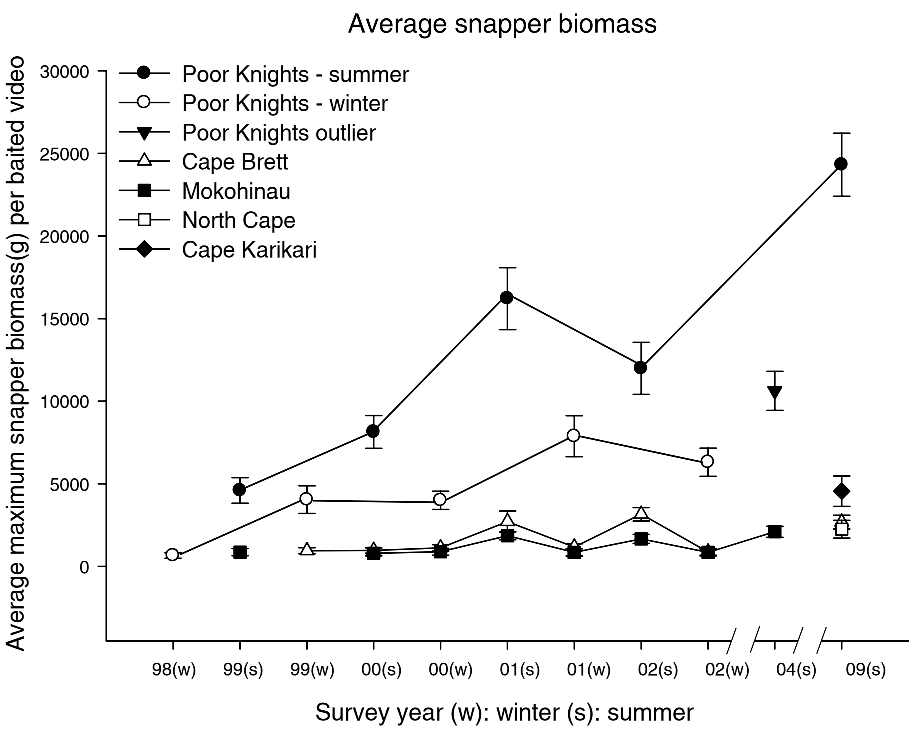


Figure 9: Average maximum snapper biomass per baited video at all locations and standard error bars.

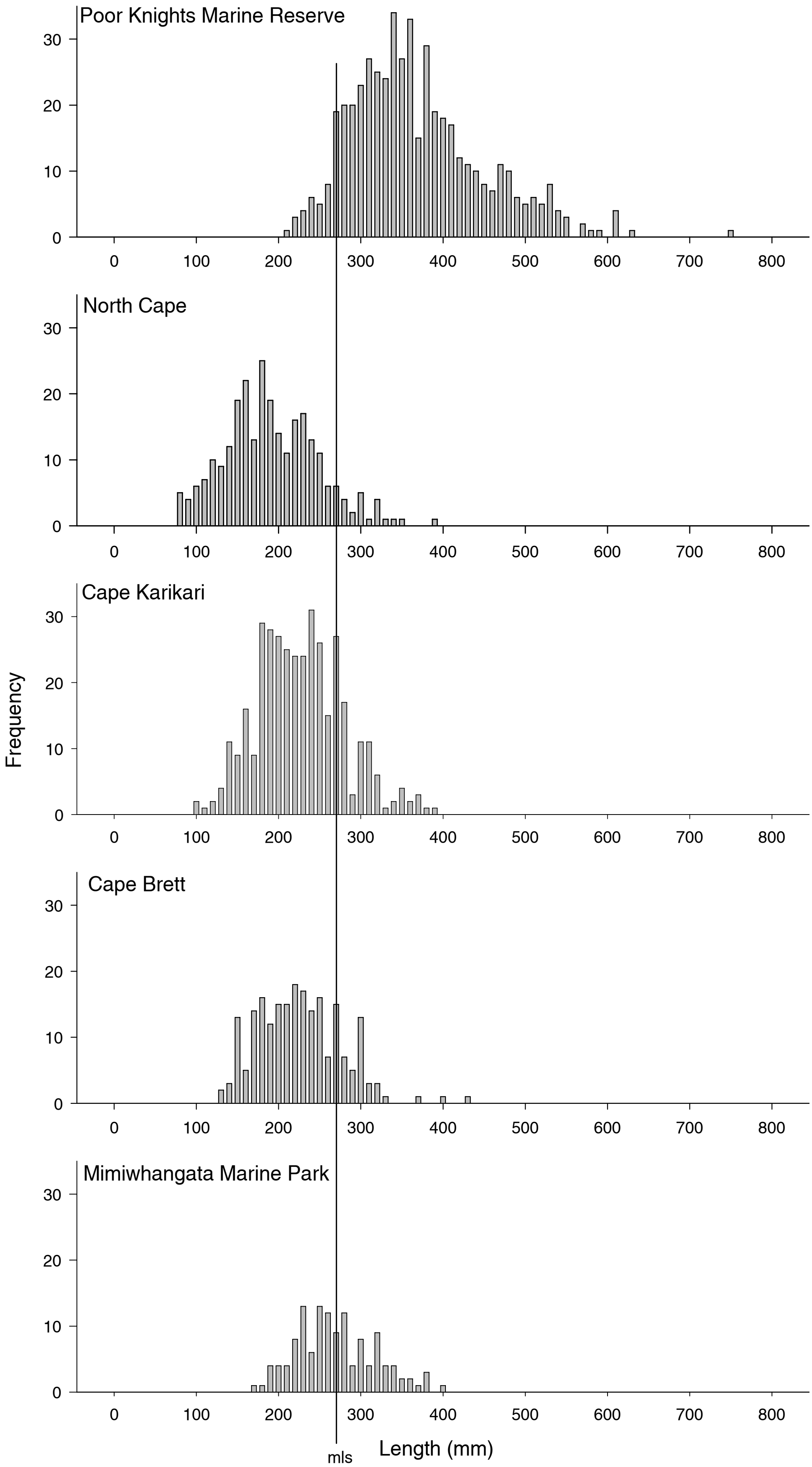


Figure 10: Size frequency distributions of snapper in 2009. The line marked mls indicates the minimum legal recreational size limit of 270 mm.

Snapper lengths were converted into biomass (Taylor & Willis 1998) using the video frame with the maximum number of snapper. Biomass per baited video was then averaged to give the average maximum snapper biomass. Snapper biomass at each location for 2009 is shown in figure 11. This figure illustrates the large significant difference in snapper biomass between the Poor Knights Islands Marine Reserve and fished reference sites at Cape Brett, Cape Karikari and North Cape. Snapper biomass at North Cape was 9.3% that of the Poor Knights, Cape Karikari snapper biomass was 18.7% that of the Poor Knights and snapper biomass at Cape Brett was 11.1% that of the Poor Knights.

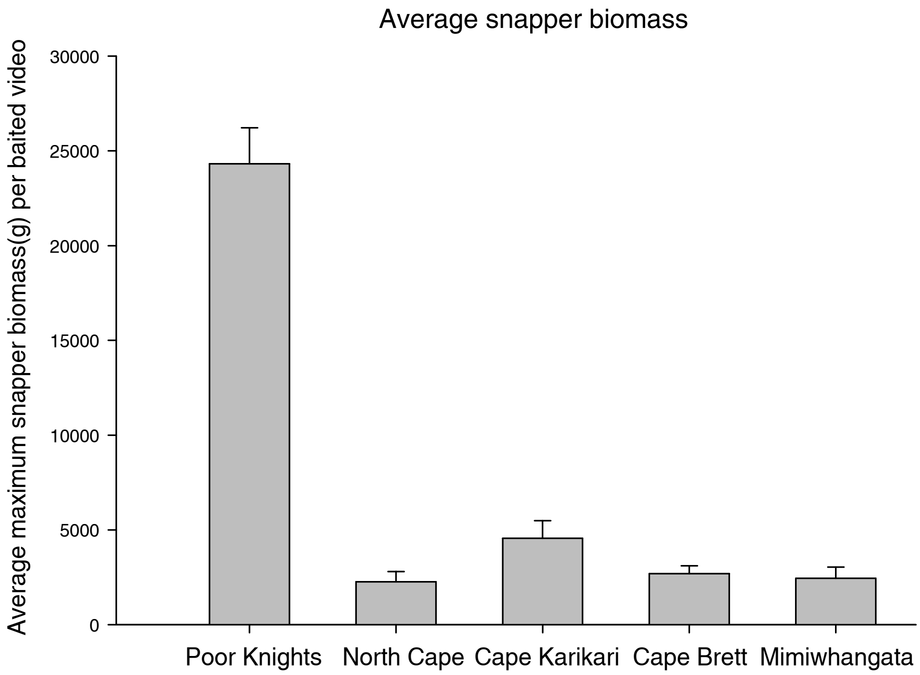


Figure 11: Average maximum snapper biomass per baited video in 2009 and standard error bars.

Changes in biomass over time at all monitoring locations are illustrated in figure 9. In the winter sample of 1998, while the Poor Knights was still open to some forms of fishing, average maximum snapper biomass was 663 g per baited video. By the winter of 2002 after four years of no take protection average maximum snapper biomass was 6.3 kg a significant increase of 952%. Average maximum snapper biomass in the first summer sample (1999) was 4.6 kg. After more than ten years of protection snapper biomass in the summer of 2009 was 24.3 kg. This is an increase in biomass of 528% from the first summer sample taken in 1999. In contrast snapper biomass at the fished reference locations of Cape Brett and the Mokohinau Islands has remained relatively unchanged with only a small increase over the ten year sampling period. At Cape Brett in the summer of 1999 snapper biomass was 955 g; in 2001 average maximum snapper biomass increased to 2.7 kg and has since remained consistent. At Cape Brett in 2009 (summer) maximum snapper biomass was 2.7 kg. At the Mokohinau Islands snapper biomass changed from 222 g in the summer of 1999 to 333 g in the summer of 2004.

Kingfish counts have fluctuated over the sampling period however summer counts have increased from 0.09 to 0.84 after ten years of protection. This is an increase from approximately one kingfish every ten camera drops to over eight kingfish every ten drops. In strong contrast to the Poor Knights, kingfish counts at Cape Brett, Cape Karikari and the Mokohinau Islands were much lower. North Cape had the highest average kingfish count second only to the Poor Knights. Average baited video counts for pigfish, sandagers wrasse, porae, northern scorpion fish, trevally and leatherjacket are illustrated in figures 13–17.

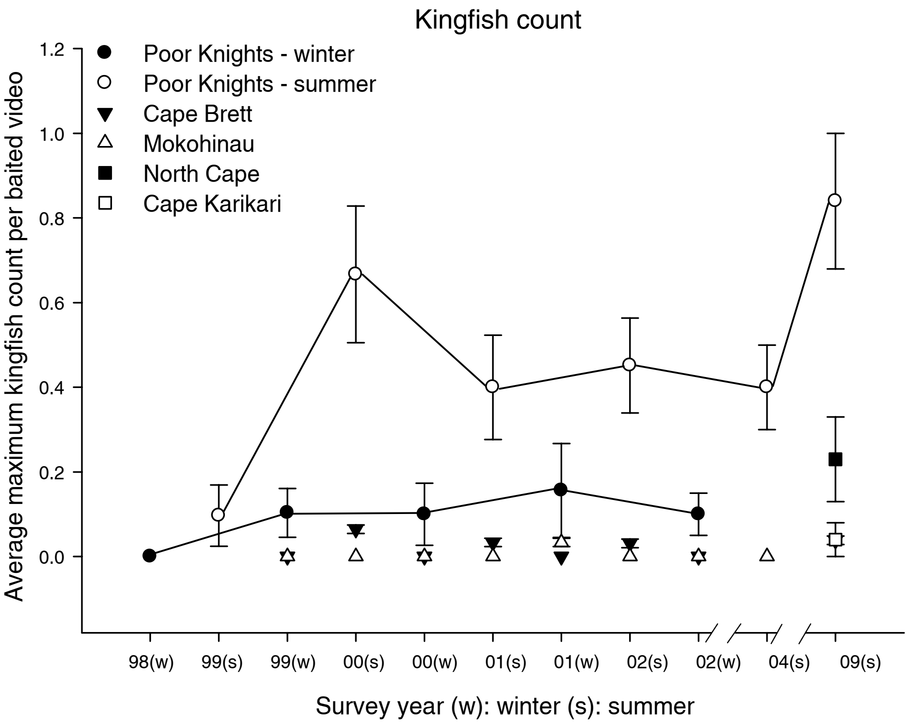


Figure 12: Average maximum kingfish counts per baited video at the Poor Knights and standard error bars.

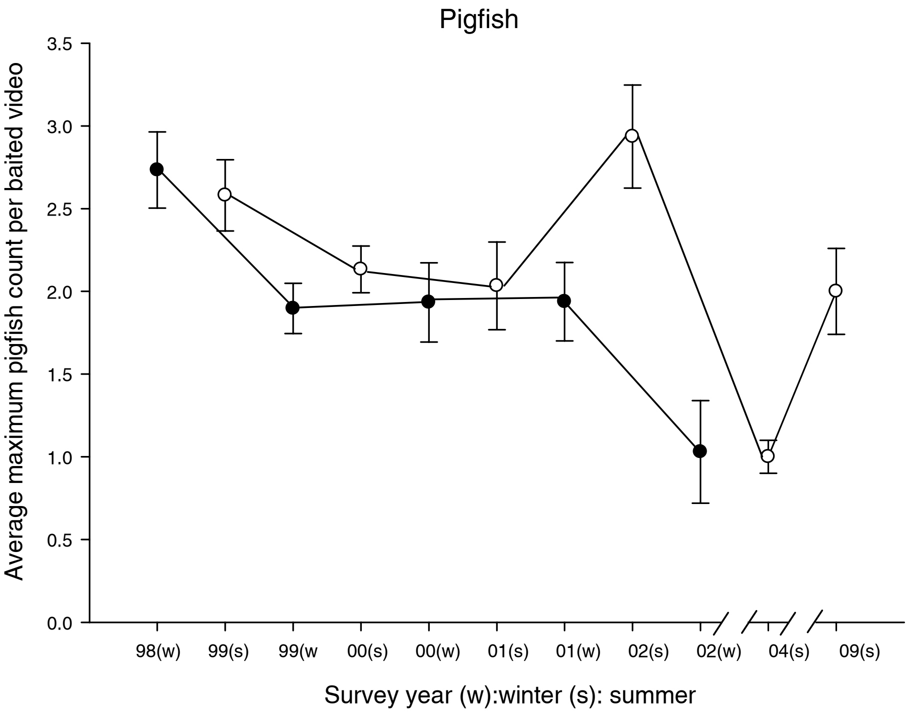


Figure 13: Average maximum pigfish counts per baited video at the Poor Knights and standard error bars.

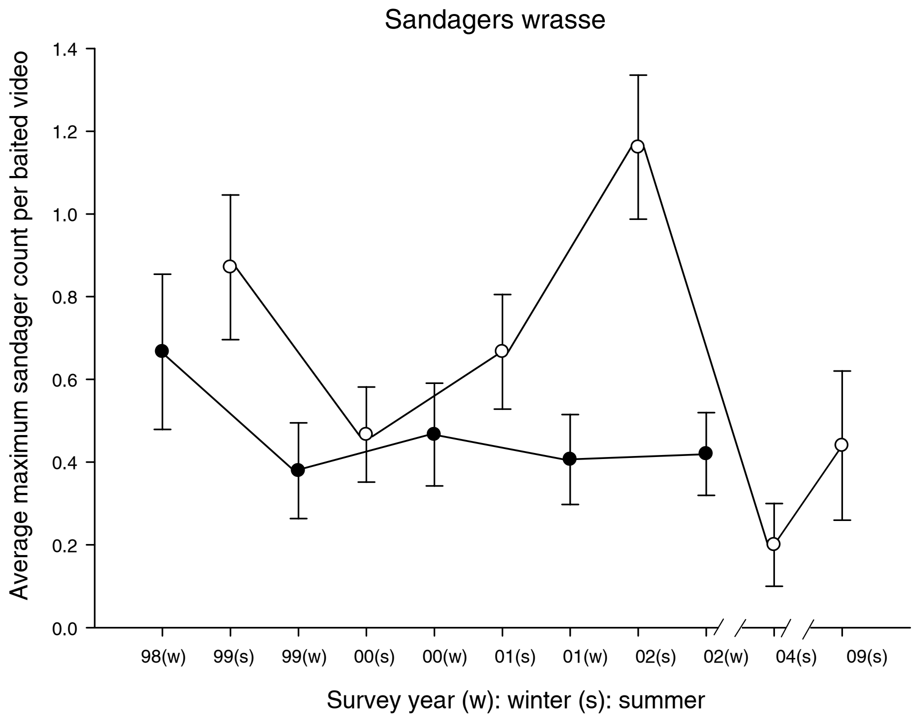


Figure 14: Average maximum sandagers wrasse counts per baited video at the Poor Knights and standard error bars.

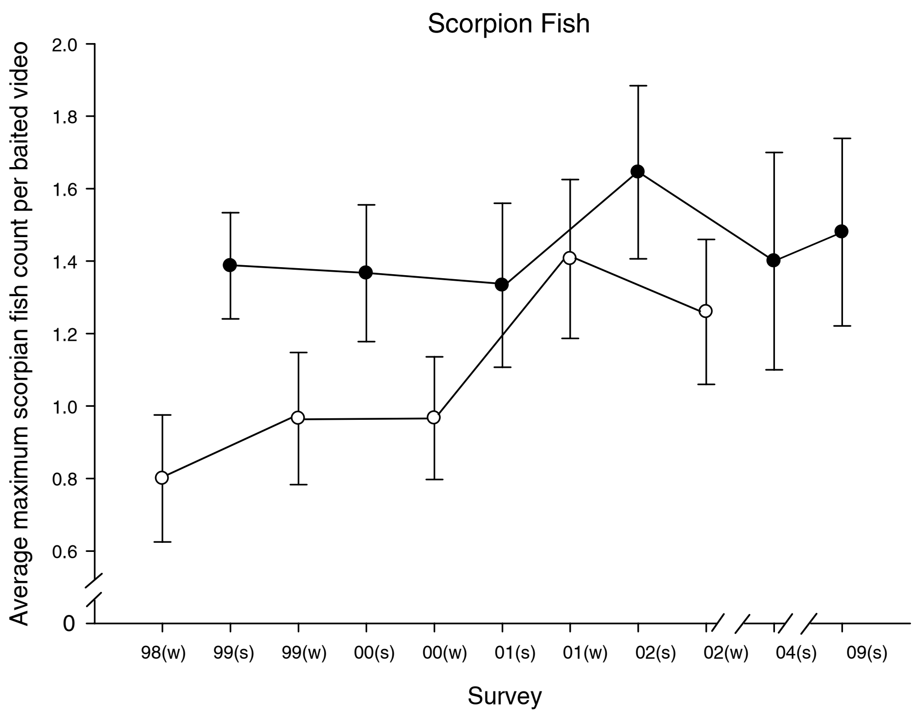


Figure 15: Average maximum scorpion fish counts per baited video at the Poor Knights and standard error bars.

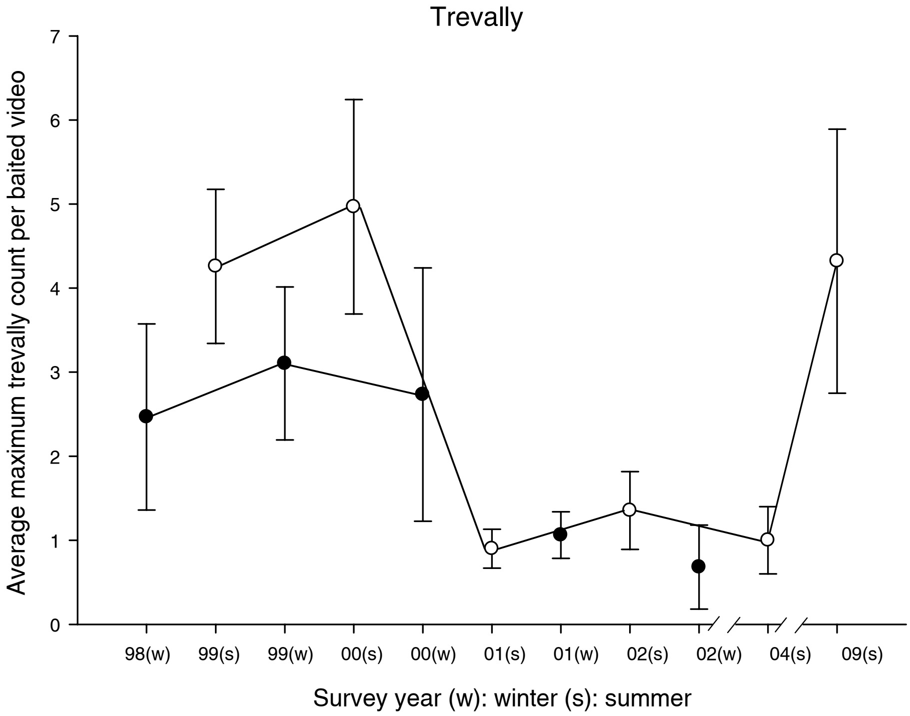


Figure 16: Average maximum trevally counts per baited video at the Poor Knights and standard error bars.

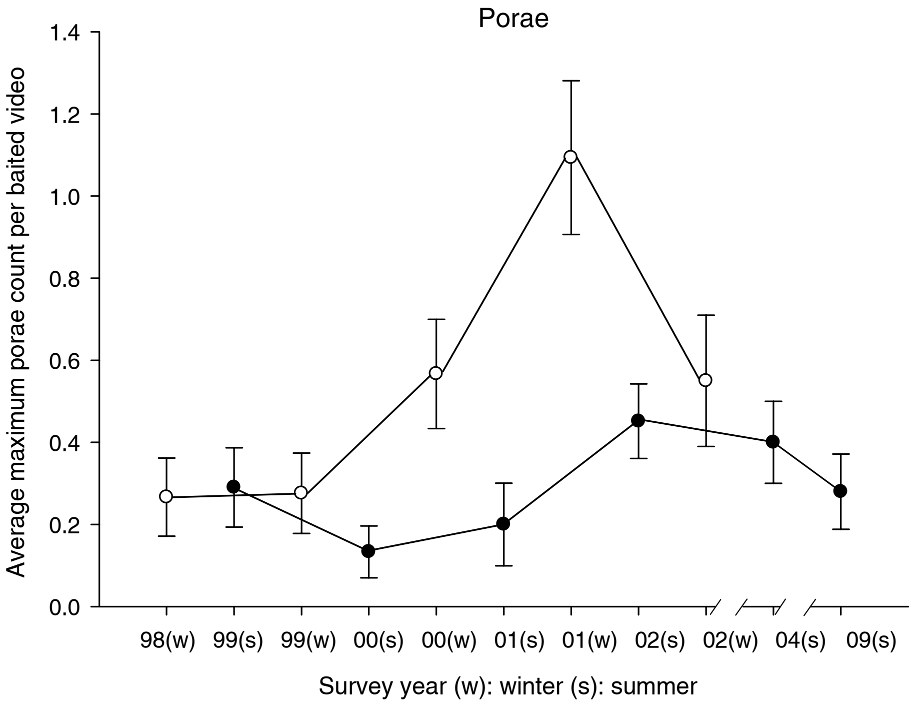


Figure 17: Average maximum porae counts per baited video at the Poor Knights and standard error bars.

MIMIWHANGATA

Twenty-nine baited video drops were completed at Mimiwhangata, 16 inside the marine park and 13 outside. The habitats that were sampled inside and outside the marine park were predominantly sand or sand/gravel adjacent to reef.

Average baited video snapper abundances are graphically illustrated in figure 18. Statistical analysis using ratios to compare snapper abundances inside and outside the marine park are given in table 6. Snapper abundances were not significantly different inside versus outside the Mimiwhangata Marine Park in 2009. The average maximum number of snapper (ams) within the marine park was 6.88 (ams) ± 1.29 and the average maximum number of snapper outside the marine park was 5.42 (ams) ± 1.51. This result is consistent with baited video samples taken in 2002 and 2003.

TABLE 6: A STATISTICAL COMPARISON OF SNAPPER ABUNDANCE INSIDE VERSUS OUTSIDE THE MIMIWHANGATA MARINE PARK IN 2009 USING RATIOS. P – VALUES < 0.05 SHOW A SIGNIFICANT DIFFERENCE.

STATUS	RATIO	UPPER CL	LOWER CL	P - VALUE
Marine Park	0.983734	1.498128	0.645961	0.939
Not Marine Park	1.0	1.0	1.0	1.0

A box and whisker plot of snapper fork lengths is given in figure 19. Snapper lengths were found to be statistically ($p = 0.015$) smaller inside the Marine Park than outside of the Marine Park. Mean snapper length inside the marine park was 265 mm ± 4. Mean snapper length outside of the marine park was 285mm ± 8.

Average maximum snapper biomass within the Mimiwhangata Marine is compared with the Poor Knights Islands Marine Reserve, North Cape, Cape Karikari and Cape Brett in figure 11. Average maximum snapper biomass per baited video in 2009 at the Poor Knights Islands Marine Reserve was 24 kg ± 2, 10× greater than the average snapper biomass in the Mimiwhangata Marine Park which was 2.4 kg ± 0.6 kg. Average maxim snapper biomass inside the Mimiwhangata Marine Park was not significantly different to all sampled locations open to fishing.

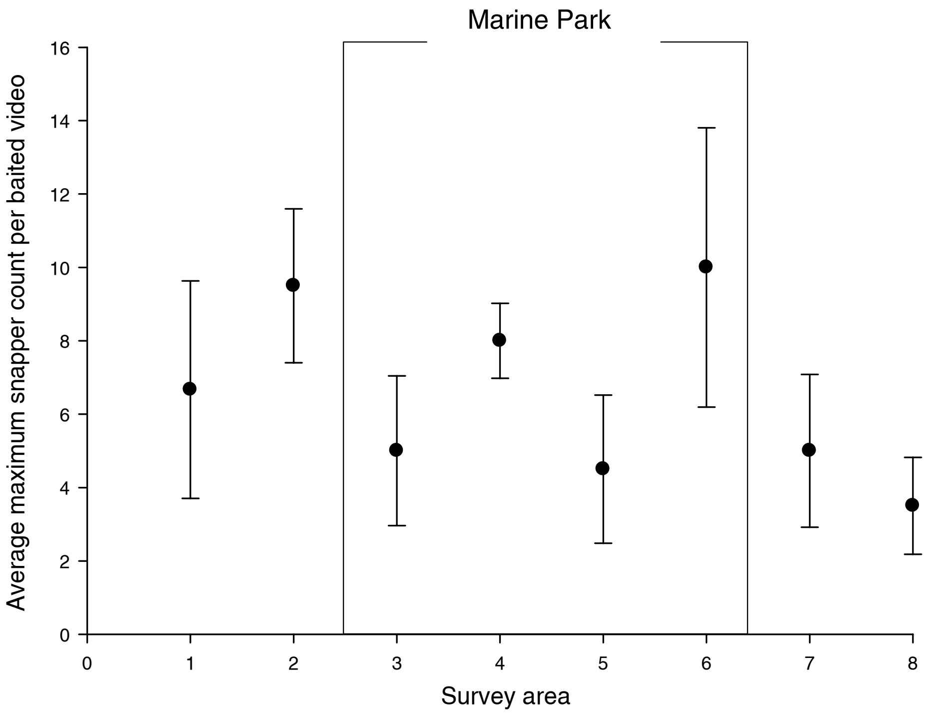
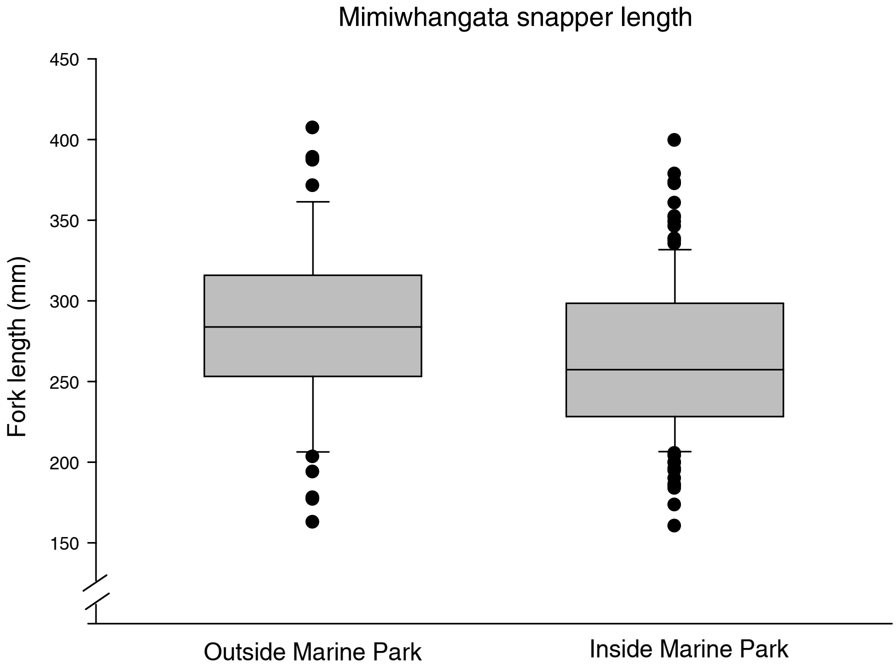


Figure 18: Average maximum number of snapper per baited video inside and outside the Mimiwhangata Marine Park in 2009 and standard error bars. Survey areas 1, 2, 7 and 8 were outside of the marine park. Survey areas 3, 4, 5 and 6 were inside the marine park.

Figure 19: A Box and whisker plot of snapper length inside versus outside the Mimiwhangata Marine Park in 2009. The lower boundary of the box indicates the 25th percentile; the line in the box represents the median, and the upper boundary of the box the 75th percentile. The whiskers above and below the box indicate the 90th and 10th percentiles and the black circles indicate outliers.



Discussion

POOR KNIGHTS ISLANDS MARINE RESERVE

The Poor Knights Islands Marine Reserve has now been fully protected from fishing for over 11 years. This investigation reinforces previous findings that snapper have increased in abundance and size since monitoring began in 1998 when the area was fished recreationally. This data also supports observations by divers who report a massive increase in the numbers and size of snapper at the Poor Knights since the marine reserve was made fully no take. In strong contrast to the Poor Knights Marine Reserve snapper abundances at North Cape, Cape Karikari and Cape Brett were much lower and fish over minimum legal size were relatively rare. North Cape, Cape Brett and Cape Karikari are partially isolated locations and, it could be expected that fish populations at these locations were protected to some degree, by their isolation. Calculations based on length measurements and abundance show that there is a large difference in biomass between recorded snapper populations at the Poor Knights and recorded snapper at North Cape, Cape Karikari and Cape Brett. This large difference is probably a result of sustained commercial and recreational fishing pressure.

Recorded snapper abundances at the Poor Knights have been highly variable between seasons. Abundances recorded in summer samples are consistently higher than in winter samples. This seasonal pattern has also been recorded in other marine reserves throughout New Zealand (Willis et al. 2003, Sivaguru 2007). Patterns in the seasonal catchability of snapper are well known by fishers in Northland with particular areas fishing differently in the summer months than winter months theoretically explained by some fish moving from deeper to shallower waters in summer. Baited video recordings may therefore be affected by regional variations in the seasonal movement of snapper to inshore environments. However (Willis et al. 2001, Parsons et al. 2003) found that tagged snapper within a marine reserve can have small home ranges and strong association with particular reefs. The snapper population at the Poor Knights probably consists of both resident snapper and seasonal migrants however a tagging study is required to verify this further. Snapper abundances in winter recordings have been much less variable than summer and may reflect fish that are resident year round. A winter sample at the Poor Knights will provide more insight into whether the winter snapper population has continued to increase in size and abundance since 2002, the last time it was monitored.

At the Poor Knights initial rapid recovery of snapper populations following implementation of no-take status were due to an influx in migratory fish rather than recovery of a resident population (Denny & Babcock 2003). Surveys conducted in 2009 indicate that snapper numbers may be leveling out after an initial strong increase. There was no significant difference in average snapper counts between summer 2001 and summer

2009 surveys. However, the apparent lack of change in abundance of snapper populations at the Poor Knights since 2001 may be explained by the monitoring method utilised. As fish size has increased, fish have become more competitive, excluding smaller fish from the bait (author pers obs.). Fewer, larger fish are then recorded within the cameras field of view. Average snapper size in 2009 was the highest ever recorded at the Poor Knights. Biomass calculations provide an alternative measure to abundance by also taking into account size. Snapper biomass at the Poor Knights has increased over the past five years indicating that snapper abundance and size may still be increasing at the Poor Knights.

The baited underwater video system used in this investigation has a limited capability to measure any further increase in the abundance of the snapper population at the Poor Knights in summer. In most locations at the Poor Knights in summer, the baited video system is saturated and is unable to measure the large biomass of fish outside the cameras field of view. Summer baited video sampling is therefore probably underestimating the size of the snapper population. This is not an issue at the fished reference locations where there are much lower numbers of fish, and competition for space around the bait does not result in fish being excluded from the video's camera frame. As a result differences in abundance estimates between the Poor Knights Islands Marine Reserve and fished reference locations may be much greater than has been indicated by data within this report, but at this stage it is not possible to know this for sure.

A simple solution to snapper saturation in summer would be to undertake a winter sample. Winter average snapper counts are much lower than summer counts and therefore are less likely to saturate the baited video system. A winter sample will help to resolve whether snapper numbers have continued to increase, or have in fact leveled out at the Poor Knights. An alternative option is a baited video system currently being used in Australia. This method is based on the stereo-video systems developed by Mark Shortis and Euan Harvey for the Department of Conservation for use in Fiordland (Harvey & Shortis 1996). This system uses two horizontally facing cameras giving a much greater field of view than the baited video system employed in this study. The larger field of view enables more abundant fish populations to be captured and typically results in more species being sampled. This system may be capable of capturing the large population of snapper present at the Poor Knights over summer, but would need to be tested and compared with the method used here.

The size frequency distributions of snapper populations at the Poor Knights were much different than monitoring sites open to fishing. Large fish were rare at North Cape, Cape Karikari and Cape Brett and recorded snapper at all of the fished locations were typically small fish below the legal catch size of 270mm. In contrast to coastal reefs small fish less than 212mm were not recorded at the Poor Knights. The absence of fish less than 212mm recorded on baited video cameras at the Knights may be a result of larger fish competitively excluding smaller fish from the bait in some cases. However not all camera drops were dominated by large fish and therefore if small fish were present there is a high likelihood they would have been recorded, at least in low numbers. The absence of small fish at the Poor Knights may also be explained by its offshore location and remoteness from nursery areas. Snapper may need to reach a minimum size before they are capable of relocating from inshore nursery habitats to this offshore Island group.

Baited video recorded much higher numbers of juvenile fish at North Cape and Cape Karikari than Cape Brett. It is well known that estuarine environments provide important habitat for juvenile reef fish species and Parengarenga and Rangaunu harbours hold large abundances of juvenile fish, particularly snapper (Author pers. obs, (Morrison et al. 2009)). The proximity of sampling sites at North Cape and Cape Karikari to these nursery areas explains the high numbers of small snapper recorded on baited video. Estuarine environments throughout New Zealand have been stressed by sediment loading and pollution and this has limited their capacity to provide suitable environments for juvenile fish species (Morrison et al. 2009). Rangaunu and Parengarenga have been found to be amongst the most pristine estuarine systems on the east coast (Morrison et al. 2009) and may be contributing a high proportion of fish recruits to east coast marine habitats. Parsons et al. (2003) found that snapper can have small home ranges and therefore the establishment of marine reserves within close proximity to Parengarenga and Rangaunu harbours may protect high numbers of resident juvenile fish species as well as seasonally migrating mature fish.

Investigations of marine reserves throughout New Zealand have demonstrated that popular fished species such as snapper, blue cod (*Parapercis colias)* and crayfish (*Jasus edwardsii)* can recover following protection and cessation of fishing (Willis et al. 2003, Denny & Shears 2004, Denny et al. 2004, Shears et al. 2006, Davidson et al. 2007, Freeman 2008). Recovery of these high level predators in some cases has been linked to community level changes in other species. For example long term changes at monitoring sites in the Cape Rodney to Okakari Point (Leigh) Marine Reserve provide strong evidence that the recovery of previously fished predators (snapper & crayfish) can result in more natural urchin populations and a recovery of algal forests (Babcock et al. 1999, Shears 2003, Shears & Babcock 2003). To investigate potential community-level effects caused by the increase in snapper at the Poor Knights, subtidal reef communities were surveyed in 2006 and compared with data from the first year of no-take protection (1999) (Shears 2007). For algal species composition, algal community structure, and sessile benthic communities there was no significant difference between the two surveys. This suggests that large increases in the snapper population have not yet had an effect on sea urchin abundance (Shears 2007). Further sampling over sufficient time periods, along with dietary and habitat use studies, are necessary to resolve any affects on benthic species and habitats at the Poor Knights Islands Marine Reserve.

For carnivorous species other than snapper it is more difficult to determine trends in abundance from the baited video data. These species are present at all locations in relatively low numbers and the data is often highly variable between locations, sites and years. At the Poor Knights, winter kingfish counts have not changed significantly since the marine reserve was established. However in summer samples, average maximum kingfish counts have increased from 1998 to 2009. Results are still highly variable and further sampling will reveal a clearer result for kingfish at the Poor Knights. The downward facing baited video camera used in this study also has a limited capability to capture large schools of kingfish and is probably not a good method for monitoring kingfish abundance. Horizontally facing cameras may provide more comprehensive information about the effects of no take protection on kingfish abundance at the Poor Knights.

Pigfish and sandagers wrasse may have dropped off in numbers at the Poor Knights since monitoring began in 1998. A reduction in numbers may be a result of increased competition with snapper for food or increased direct predation pressure. An alternative theory is that declines may be due to natural mortality after a recruitment pulse in 1998 due to warmer sea surface temperatures (Denny & Shears 2004). However differences in abundance counts have not changed much over the sampling period and a longer time series of data is required to resolve the effect of the marine reserve on these two species. Porae and northern scorpion fish are both vulnerable to some forms of fishing. However, neither species has increased significantly in abundance at the Poor Knights. Baited video is probably not an effective monitoring method for trevally because of this species' tendency to school, resulting in highly variable data.

MIMIWHANGATA

Within the Mimiwhangata Marine Park recreational fishing restrictions have existed for 26 years and commercial long lining for snapper has now been prohibited for 17 years. We therefore had an opportunity to test the effectiveness of partial protection over a long time scale. Baited underwater video sampling in this investigation indicates that there is no significant difference in the abundance of snapper inside versus outside the marine park at Mimiwhangata. The average length of snapper was also smaller inside than outside of the marine park. This finding is consistent with previous monitoring work at the Mimiwhangata Marine Park (Denny & Babcock 2002, Usmar et al. 2003). Snapper biomass within the Mimiwhangata Marine Park was compared with snapper biomass within normally fished areas of the adjacent coastline and with North Cape, Cape Brett and Cape Karikari. These comparisons indicated that snapper biomass within the marine park was not significantly different from other areas open to fishing. This study concludes that Mimiwhangata Marine Park fishing restrictions are not achieving their goal of protecting biodiversity while allowing for limited recreational take.

It is currently unknown the degree to which special fishing regulations at Mimiwhangata are affecting recreational fishing activities within the marine park. When Mimiwhangata was sampled in 2009 a number of recreational vessels were observed fishing within the marine park using illegal methods (pers. obs). Signage at popular boat launching areas around Mimiwhangata such as Oakura is limited and some recreational fishers targeting Mimiwhangata are using standard fishing equipment and are not complying with the Marine Park regulations (pers. obs). Those who are aware of the regulations and choose to follow them have a range of options to capture popular table species. Stray lining is an effective method of catching snapper and can be used freely within the marine park. Fishers can also weight their lines with large pieces of bait to get their hooks into the habitats they are targeting. Fishing regulations existing at Mimiwhangata are probably not limiting recreational fishers' ability to target species such as snapper which may have limited the recovery of this species within the marine park.

Conclusions

Some marine reserves in New Zealand have consistently demonstrated their effectiveness at protecting fish, lobsters and shellfish previously targeted by fishermen (Shears & Babcock 2003, Davidson & Richards 2005b, Haggitt & Mead 2006, Sivaguru 2007, Freeman 2009). In New Zealand's oldest marine reserve at Leigh, increased abundances of fish have been shown to affect entire reef systems due to close links between fish, algae and invertebrates (Babcock et al. 1999, Shears & Babcock 2003). The Poor Knights Islands Marine Reserve has resulted in the effective protection of an abundant population of large snapper. A longer time series of data, using a range of monitoring methods and manipulative experiments are required to resolve the effects of protection for other species of reef fish, algal and invertebrate communities.

In New Zealand there is currently a lot of interest in the use of partial protection regulations to protect marine ecosystems while allowing for harvesting of some species. Theoretically we may be able to minimize the impacts on existing users while maximizing biodiversity objectives. Snapper are the most heavily fished species in northern New Zealand and where marine reserves are enforced snapper populations have rapidly recovered (Kelly 2000, Willis et al. 2003). Partial protection regulations have existed at the Mimiwhangata Marine Park for 17 years and therefore if the regulations were effective we could reasonably expect snapper to be more abundant and larger inside the protected area. However baited video monitoring showed that snapper were not larger and more abundant even after 17 years of partial protection. This study therefore supports previous conclusions that while they may protect some components of biodiversity, partial protection mechanisms can be ineffective conservation tools for heavily targeted species such as snapper.

New Zealand has committed to creating a viable network of marine protected areas. The Poor Knights, North Cape, Cape Karikari, Cape Brett, Mokohinau and Mimiwhangata data sets provide the ability to evaluate the performance of marine protected areas against fished areas throughout northern New Zealand. This information will be vital to guide the establishment and management of successful marine protected area networks in the future.

Recommendations

1. Diver observations of the baited video in summer months indicated that in many locations this monitoring method may be saturated and unable to capture the large numbers of fish outside the cameras field of view. A leveling out of snapper abundance may be a reflection of the limitations of the baited video method rather than a real trend.

A winter baited video sample will help to resolve whether snapper numbers have continued to increase, or have in fact leveled out at the Poor Knights.

2. Stereo baited video could be trialed at the Poor Knights to determine whether the downward facing baited videos used presently are reaching saturation point. Paired standard baited video and stereo baited video could be used to determine if the stereo system is capable of measuring a greater range of relative abundance than the standard baited video. This information is required to assess whether the current method is an appropriate method for the future or if an alternative method is required to monitor the long term response of reef fish to no take protection.
3. Little is known about the effects such a large increase in predatory biomass will have on other components of the ecosystem at the Poor Knights. Further investigations into habitat utilisation and the diet of snapper are necessary to resolve any potential effects on benthic species and habitats at the Poor Knights Islands Marine Reserve.

Acknowledgements

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More recently many people have helped to continue this project and I would like to thank; Alan Fleming for going above and beyond to skipper and crew sometimes in arduous conditions and Fiona Watson and Renee Attwood for analysing the video footage and methodically inputting all of the data. Thanks to all those that helped with the multitudes of tasks that have been necessary to complete this project; Amy Macdonald, Rolf Fuchs, Pete Davis, Kaye Seymour, Keith Hawkins, Piet Nieuwland, Andrew Macdonald, Sue Quigg, Maree Attwood, Christian Macdonald, Leon Candy, Patrick Whaley, Lester Bridson, Irene Petrov, Willy Macrae, Tim Langlois, Hilary Aikman, Vince Kerr. Thanks to Nick Shears and Debbie Freeman for reviewing the draft version.

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Appendix 1:

MONITORING SITE INFORMATION

TABLE 7: CAPE KARIKARI MONITORING SITE INFORMATION 2009

LATITUDE	LONGITUDE	DEPTH (m)	DATE	TEMPERATURE	HABITAT
-34.76620	173.34202	28.38	24.3.09	20.4	reef
-34.76758	173.34249	26.8	24.3.09	20.4	sand
-34.77097	173.34384	15.82	24.3.09	20.5	sand
-34.77634	173.35055	13.91	24.3.09	20.4	sand
-34.77987	173.35196	14.77	24.3.09	20.4	sand
-34.78207	173.35642	13.93	24.3.09	20.4	sand
-34.78443	173.35957	13.3	24.3.09	20.4	mixed
-34.78667	173.36577	16.93	24.3.09	20.4	sand
-34.78829	173.37309	16.35	24.3.09	20.4	mixed
-34.79180	173.37899	12.17	24.3.09	20.4	mixed
-34.78838	173.38364	19.41	24.3.09	20.4	mixed
-34.78279	173.38618	30.41	24.3.09	20.4	sand
-34.78378	173.37910	28.86	24.3.09	20.4	sand
-34.78205	173.37299	31.34	24.3.09	20.4	mixed
-34.76757	173.35454	32.67	25.3.09	20.4	mixed
-34.76610	173.34700	36.2	25.3.09	20.4	mixed
-34.77064	173.35605	30	25.3.09	20.4	reef
-34.77484	173.35851	29.88	25.3.09	20.4	mixed
-34.77760	173.36338	35.73	25.3.09	20.4	sand
-34.78101	173.36623	30.41	25.3.09	20.4	sand
-34.78279	173.39461	26.12	25.3.09	20.4	sand
-34.78475	173.39345	22.41	25.3.09	20.4	sand
-34.79052	173.39092	16.81	25.3.09	20.4	mixed

TABLE 8: CAPE BRETT MONITORING SITE INFORMATION 2009

LATITUDE	LONGITUDE	DEPTH (m)	DATE	TEMPERATURE	HABITAT
-35.20156	174.28723	37.83	15.4.09	19.8	mixed
-35.1988	174.28913	41.01	15.4.09	19.8	sand
-35.20147	174.29333	25	15.4.09	19.8	mixed
-35.20025	174.29744	21.63	15.4.09	19.9	mixed
-35.19787	174.30096	26	15.4.09	19.8	sand
-35.19392	174.30223	25.55	15.4.09	19.9	reef
-35.19434	174.29923	24.17	15.4.09	19.9	mixed
-35.19328	174.29218	34.23	15.4.09	19.9	mixed
-35.18412	174.29551	25.92	15.4.09	19.8	mixed
-35.17978	174.29652	33.02	15.4.09	19.8	reef
-35.18017	174.30697	34.7	15.4.09	19.9	mixed
-35.18057	174.31583	27.15	1.5.09	17.9	sand
-35.18148	174.32500	22.49	1.5.09	17.9	mixed
-35.17762	174.32866	22.72	1.5.09	18	mixed
-35.17123	174.32992	19.99	1.5.09	18	mixed

LATITUDE	LONGITUDE	DEPTH (m)	DATE	TEMPERATURE	HABITAT
-35.16598	174.33772	30.83	1.5.09	18.1	reef
-35.17485	174.33539	27.07	1.5.09	18.1	reef
-35.18033	174.33444	27.01	13.5.09	18	mixed
-35.18741	174.33904	27.21	13.5.09	18	mixed
-35.19063	174.34213	31.42	13.5.09	18	mixed
-35.19499	174.34246	35.32	13.5.09	18	mixed
-35.19244	174.33718	21.36	13.5.09	18	mixed
-35.19377	174.33438	25.22	13.5.09	18	mixed
-35.19921	174.32838	33.62	13.5.09	18	mixed
-35.19572	174.32037	17.07	13.5.09	18	mixed
-35.20379	174.31910	32.34	13.5.09	18	mixed

TABLE 9: NORTH CAPE MONITORING SITE INFORMATION 2009

LATITUDE	LONGITUDE	DEPTH (m)	DATE	TEMPERATURE	HABITAT
-34.42600	E173.01999	10.63	26.3.09	20.8	mixed
-34.42574	E173.02481	14.61	26.3.09	20.8	sand
-34.42797	E173.02713	18.59	26.3.09	20.8	sand
-34.42819	E173.03018	21.61	26.3.09	20.8	sand
s	e	?	26.3.09	20.8	mixed
-34.42326	E173.02765	14.1	26.3.09	20.8	sand
-34.42774	E173.03757	23.74	26.3.09	20.8	mixed
-34.42008	E173.03338	16.23	26.3.09	20.8	sand
-34.41762	E173.03833	15.55	26.3.09	20.8	sand
-34.41622	E173.04438	15.82	26.3.09	20.8	sand
-34.42248	E173.03914	21.12	26.3.09	20.8	mixed
-34.41942	E173.04472	20.93	26.3.09	20.8	mixed
-34.42563	E173.04344	26.17	26.3.09	20.8	mixed
-34.42219	E173.04702	24.2	26.3.09	20.8	mixed
-34.42233	E173.05215	24.24	26.3.09	20.8	mixed
-34.41799	E173.05167	18.63	26.3.09	20.8	reef
-34.42529	E173.05009	28.32	27.3.09	20.8	mixed
-34.42660	E173.04713	30.21	27.3.09	20.8	mixed
-34.42833	E173.04295	31.42	27.3.09	20.8	mixed
-34.42972	E173.04864	33.27	27.3.09	20.8	mixed
-34.43074	E173.04106	30.43	27.3.09	20.8	mixed
-34.43193	E173.04573	34.46	27.3.09	20.8	sand
-34.43190	E173.03780	29.66	27.3.09	20.8	mixed
-34.43338	E173.04633	37.11	27.3.09	20.8	mixed
-34.43246	E173.03318	26.68	27.3.09	20.8	sand
-34.43200	E173.02728	21.28	27.3.09	20.8	sand

TABLE 10: MIMIWHANGATA MONITORING SITE INFORMATION 2009

LATITUDE	LONGITUDE	DEPTH	DATE	AREA	HABITAT
-35.48782	E174.46312	10.39	7.5.09	8	mixed
-35.48219	E174.46443	17.00	7.5.09	8	sand
-35.47682	E174.45617	13.71	7.5.09	8	sand
-35.47383	E174.46403	19.99	7.5.09	8	mixed
-35.46867	E174.45800	16.91	7.5.09	7	mixed
-35.46416	E174.45283	14.39	7.5.09	7	mixed

LATITUDE	LONGITUDE	DEPTH	DATE	AREA	HABITAT
-35.46417	E174.44651	10.24	7.5.09	7	mixed
-35.45918	E174.44366	13.61	7.5.09	7	sand
-35.45268	E174.43734	13.20	7.5.09	6	sand
-35.44850	E174.43502	12.71	7.5.09	6	sand
-35.43999	E174.44186	18.49	7.5.09	6	mixed
-35.43536	E174.45118	25.08	8.5.09	6	mixed
-35.43068	E174.44733	16.93	8.5.09	5	mixed
-35.43082	E174.43729	9.50	8.5.09	5	mixed
-35.42647	E174.45186	24.16	8.5.09	5	reef
-35.41926	E174.45423	31.20	8.5.09	5	reef
-35.41830	E174.44810	29.25	8.5.09	4	reef
-35.41784	E174.44298	28.92	8.5.09	4	mixed
-35.41769	E174.42513	27.05	8.5.09	4	mixed
-35.42048	E174.42440	20.44	8.5.09	4	mixed
-35.42383	E174.41419	20.48	8.5.09	3	mixed
-35.43017	E174.41716	4.70	9.5.09	3	sand
-35.42819	E174.40165	17.20	9.5.09	3	sand
-35.42534	E174.39199	17.57	9.5.09	3	sand
-35.42952	E174.38134	11.31	9.5.09	2	sand
-35.43069	E174.37420	8.13	9.5.09	2	sand
-35.42519	E174.37583	10.98	9.5.09	2	sand
-S35.41969	E174.37970	16.50	9.5.09	1	sand
-35.41469	E174.37911	15.83	9.5.09	1	mixed

TABLE 11: POOR KNIGHTS MONITORING SITE INFORMATION 2009

LATITUDE	LONGITUDE	DEPTH(m)	NAME	DATE	TEMPERATURE	HABITAT
-35.48126	E174.73388	36.43	Rikoriko	18.3.09	21.4	sand
-35.47631	E174.73587	28.57	The Labrynth	18.3.09	21.8	sand
-35.47131	E174.73385	23.44	The Gap	18.3.09	21.2	sand
-35.46873	E174.73353	41.99	Cairneys rock	18.3.09	21.4	reef
-35.46730	E174.73592	31.99	Maomao arch	18.3.09	21.5	sand
-35.46409	E174.73616	42.03	Skull Bay	18.3.09	21.7	sand
-35.46116	E174.73433	29.31	Shag Bay	18.3.09	21.7	sand
-35.47531	E174.73653	18.80	Nursery Cove	18.3.09	21.6	sand
-35.45803	E174.73268	38.28	Middle Arch	19.3.09	22.2	reef
-35.45528	E174.73136	46.16	S Cleaner Fish	19.3.09	21.5	sand
-35.48661	E174.74455	34.09	Aorangaia Is.	19.3.09	21.4	reef
-35.48921	E174.74179	17.73	Southern Arch	19.3.09	21.3	reef
-35.48985	E174.73949	31.24	Chris's Area	19.3.09	21.4	sand
-35.48937	E174.73848	29.00	Ngaio rock	19.3.09	21.4	sand
-35.47649	E174.74248	22.84	Matts Crack	20.3.09	21.2	reef
-35.47394	E174.74099	26.35	West Bartles	20.3.09	21.3	reef
-35.47048	E174.73860	38.46	Arch Rock	20.3.09	21.2	reef
-35.46186	E174.74284	37.99	RockLilley I	20.3.09	21.2	mixed
-35.45583	E174.74215	40.60	Cave Bay	20.3.09	20.9	reef
-35.47850	E174.73605	29.26	The Gardens	20.3.09	21.4	sand
-35.48482	E174.74437	36.12	N Frasers Bay	21.3.09	21.4	reef
-35.48658	E174.74187	28.14	S Frasers Bay	21.3.09	21	sand
			Annes Rock	21.3.09	21.4	reef
-35.45170	E174.73227	37.74	N Cleanerfish	21.3.09	20.8	reef
-35.44813	E174.73230	36.31	Northern Arch	21.3.09	21.4	reef