Mimiwhangata Marine Park Monitoring Report 2003

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Executive Summary

The Mimiwhangata Marine Park in northland, New Zealand, is an area closed to commercial fishing but open to most forms of recreational fishing. The effectiveness of the park in protecting exploited species was assessed by comparing areas within the park to adjacent fished areas. The 2003 survey included studies of both fish and crayfish populations. For fish, two survey methodologies were used; baited underwater video and underwater visual census. Snapper (*Pagrus auratus*), the most heavily targeted fish species in the region, showed no difference in abundance or size between the Marine Park and adjacent control areas. In fact the Marine Park tended to have lower mean numbers of snapper than areas open to all forms of fishing, consistent with the 2002 survey. Underwater visual census found significant variation in fish assemblages but these were not consistent with any putative or observed differences in fishing between these areas. Crayfish (*Jasus edwardsii*) were also surveyed by underwater visual census, which found that density did not differ significantly between the Marine Park and adjacent areas outside the park. The lack of any recovery by snapper or crayfish populations within the Marine Park, despite the exclusion of commercial fishers and restrictions on recreational fishing, indicates that partial closures have been ineffective as conservation tools. The data suggest fishing pressure within the Marine Park is at least as high as at other 'fished' sites.

1. Introduction

Marine Protected Areas (MPAs) have recently become a major focus in marine conservation. While much of the literature on MPAs has dealt with no-take areas, MPAs can offer many levels of protection and many afford only partial protection, allowing certain types of fishing. For example, Francour et al. (2001) found that amateur and commercial fishing was allowed in half the MPAs in the Mediterranean and Bohnsack (1997) pointed out that 99.5% of the Florida Keys Marine Sanctuary provided no protection for any species. The world's largest MPA, the Great Barrier Reef Marine Park, has many levels of zoning, most of which allow fishing of some kind and less than 5% of the area is no-take (Anon., 2002). With growing worldwide pressure to increase the level of protection afforded to marine habitats, partial fishing closures are often advocated by groups with direct fishing interests. Such partial closures are promoted as a 'compromise' solution allowing both protection and fishing (Willis and Denny, 2000).

Partial closures may reduce the impacts on by-catch. This is particularly so in areas affected by destructive fishing practices, and in such circumstances they can be quite effective (Thrush et al., 1998). Depending on the behaviour of fish and fishers, partial closures may result in reduction of incidental mortality even in hook and line fisheries. Furthermore, partial closures may benefit some species. Allowing fishing for the dominant predators on a reef may actually increase the abundance of prey species. This may be a useful technique to increase the abundance of an endangered prey species. However, the effectiveness of partial closures for either conservation or enhanced fishing for a subset of fishers has not been well evaluated. In spite of the number of MPAs worldwide, only a few studies have assessed the effects of partial protection on reef fish populations (Francour, 1994; Vacchi et al., 1998; Francour et al., 2001, Westera 2003).

The Mimiwhangata Marine Park was established in 1984 with the aim of protecting long-lived reef fish that are vulnerable to overfishing or have low reproductive rates. Commercial fishing in the Marine Park was prohibited, but this change was phased in gradually, and potting for rock lobster and longlining was permitted until 1st October 1993, since which time it has been prohibited. Recreational fishers are allowed to fish under special fisheries regulations prohibiting all nets and long-lines, however they may use unweighted single hooked lines, trolling, spearing and handpicking. Potting for rock lobsters is also permitted but restricted to one pot per person, or party, or boat. Species permitted to be caught within the Marine Park (Appendix 1), were all thought to be nomadic or pelagic at the time of the park's creation. That is, they were not considered part of the resident demersal reef fish assemblage.

The inclusion of these species was based on very limited knowledge of their biology and behaviour. Three of these species, trevally (*Pseudocaranx dentex*), snapper (*Pagrus auratus*), and kingfish (*Seriola lalandi*) are now known to be completely or partially resident on reefs. Trevally are reef-associated as juveniles, whereas adults can be found near reefs or in open water (Kingsford, 1989; Francis, 2001), snapper can become permanent residents on particular areas of reefs (Willis et al., 2001), and kingfish are largely reef associated rather than ocean pelagics (Saul and Holdsworth, 1991). All three species are targeted by both recreational and commercial fishers, but snapper are the most abundant demersal predatory fish species in northeast New Zealand and support New Zealand's most valuable commercial and inshore recreational fisheries.

A previous study at Mimiwhangata compared relative fish densities, and snapper densities in particular, with those at other coastal and offshore sites in the region (Denny and Babcock 2004). These sites included both fully fished and no-take marine reserves. Densities of snapper within the Mimiwhangata Marine Park were similar to fully fished sites (Mokohinau Islands and Cape Brett) and far lower than those in a nearby no take reserve (Poor Knights Is) (Denny and Babcock 2004). The general objective of this survey was to further evaluate the effectiveness of partial protection on the reef fish assemblages and crayfish within and around the Mimiwhangata Marine Park. More specifically this would provide a set of baseline data comparable with other recent studies of marine reserve effectiveness in northeastern New Zealand (e.g. Willis et al 2003, Kelly et al. 2000). In this survey, two different methods were used to provide quantitative estimates of fish abundance and size; underwater visual census and baited underwater video. Crayfish density and size were also estimated, independently from the fish censuses, using underwater visual census.

2. Methods

2.1 Study Areas

The Mimiwhangata Marine Park, established in 1984, is located on New Zealand's northeast coast (35°25'S, 174°26'E), extending 1 km offshore, and covering about 20 km^2 (Fig. 1). Within the Marine Park boundaries, there are a variety of habitats such as shallow and deep rocky reefs, boulder fields, sandy areas, urchin barrens, and algal turf flats. For the current survey, the Marine Park was divided into 4 areas, and these were compared with 4 control areas outside the Marine Park (two at either end of the Marine Park) to assess differences inside and outside the Marine Park (Fig. 1). The areas and sites surveyed in 2003 were the same as those surveyed in 2002 (Denny and Babcock 2004,. This sampling design has been used in numerous other studies of fish in New Zealand marine reserves (Willis and Babcock, 2000; Willis et al., 2000; Willis et al., 2003). The design has the dual advantages of ensuring reference areas are similar to reserve areas, as well as enabling the detection of any edge effects that might be related to the encroachment of fishing effects into the reserve (or alternatively cross boundary movements into or out of the Marine Park). While this sampling design has not previously been applied to crayfish, it should offer the same benefits and has been applied in this study, using the same sampling sites as for fish. Sampling was conducted during April 2003.

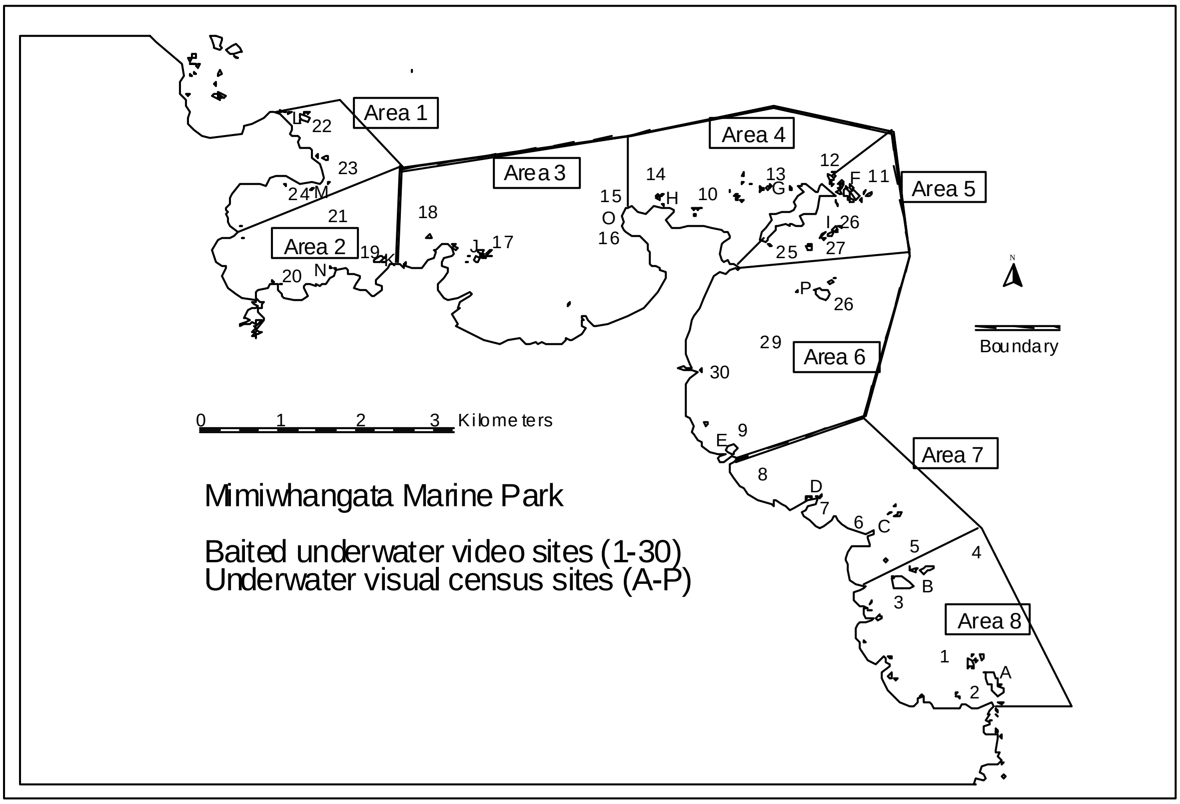

Figure 1. Map of Mimiwhangata showing the location of the baited underwater video sites (1-30) and the underwater visual census sites (A-P) in April 2003

2.2 Baited underwater video

The use of the baited underwater video technique is relatively new and allows sampling of carnivorous species that are not amenable to visual methods as well as enabling sampling at depths greater than those at which divers are able to operate (Willis and Babcock, 2000). The video system consists of a triangular stainless steel stand, with a high-resolution colour camera, positioned 1.25 metres above a bait container holding approximately 300g of pilchards, *Sardinops neopilchardus*. The baited underwater video was deployed from the research vessel to depths of up to 40 m at sites at least 1 km from diving activities (so the presence of divers would not interfere with fish responses to the bait). Each sequence was recorded for 30 min from the time the video assembly reached the bottom. A 100 m long coaxial cable connected the underwater camera to a Sony GV-S50E video monitor and 8 mm video recorder on the research vessel, which enabled the person recording to ensure the stand was upright and over suitable substratum. Four replicate video deployments were done in each of the eight survey areas (Fig. 1), except areas one and two where 3 replicate drops were done (due to logistical constraints).

Videotapes were later copied to VHS tapes for analysis and archiving. Videotapes were played back with a real-time counter, and the maximum numbers of each species of fish observed during each minute were recorded (thirty counts made during each 30-minute sequence). The lengths of snapper were obtained by digitising video images using the Sigmascan? image analysis system. Measurements were only made of those fish present when the count of the maximum number of fish of a given species in a sequence was made. While this meant that some fish moving in and out of the field of view may not have been measured, it avoided repeated measurements of the same individuals. It is likely that the use of maximum number present results in a more conservative estimate of abundance in areas of high density than at areas of low density, and therefore observed relative differences between sites are also likely to be conservative.

2.3 Fish Underwater Visual Census

Underwater visual census techniques are regularly used by researchers to quantify reef fishes, study their distribution, and to estimate their sizes (e.g. Kingsford and Battershill, 1998). The advantages of underwater visual census include the high levels of replication possible, few logistical requirements (apart from SCUBA gear), and the flexibility of being able to record other types of data *in situ*. The disadvantages include constraints of depth (less than 30 metres), high levels of inter-observer variability, diving limitations due to currents and poor underwater visibility, and bias associated with diver positive/negative species. Despite these flaws, acknowledged by most workers, underwater visual census is the best method for non-destructive surveys of a broad spectrum of fish species. In this survey, 2 sites within each of the 8 areas at Mimiwhangata were surveyed by underwater visual census (16 sites in total) (Fig. 1). Three divers recorded the numbers of all fish using 5 m x 25 m strip transects (each transect covers 125 m^2). Three replicate transects were completed at each site by each diver therefore each site covered 1125 m^2 (9 x 125 m^2). To avoid overlap divers decided which direction to swim prior to each dive. Each diver tied a fibreglass tape measure to a kelp holdfast with wire, swam out 5 metres to avoid counting species attracted to the initial activity, and preceded to swim 25 metres, counting all fish within a strip 2.5 metres either side of the diver (Willis and Denny, 2000). All divers had previous experience using this methodology.

2.4 Crayfish Underwater Visual Census

Measurement of crayfish density by scuba divers followed the methodology established by MacDiarmid (1991) and used extensively in surveys of other reserves since then (e.g. Kelly et al 2000). The method employed divers trained to visually estimate the size of crayfish and for whom precision of estimation was quantified prior to commencing the surveys (Fig. 2). This involved estimating the size of specific animals and then catching them and measuring the carapace length (CL) using vernier callipers (Fig. 2). The precision and accuracy of the estimates was excellent with the y intercept at 6.37 and the origin well within the 95 confidence interval (Fig 2). The divers also determined the sex of the crayfish. Where the divers could not be certain of the sex, usually because the pleopods or 5th pereiopods were obscured, the sex was recorded as undetermined. Crayfish transects were 50 m long by 10 m wide (500 m^2) and deployed over areas of subtidal reef within a single depth strata between 5 and 15 m. A 50 m tape measure was run out along a randomly determined compass bearing and a 5m wide area was surveyed along each side of the tape. Crayfish surveys were carried out at the same sites as for fish UVC and divers coordinated their movements to avoid overlap and interference.

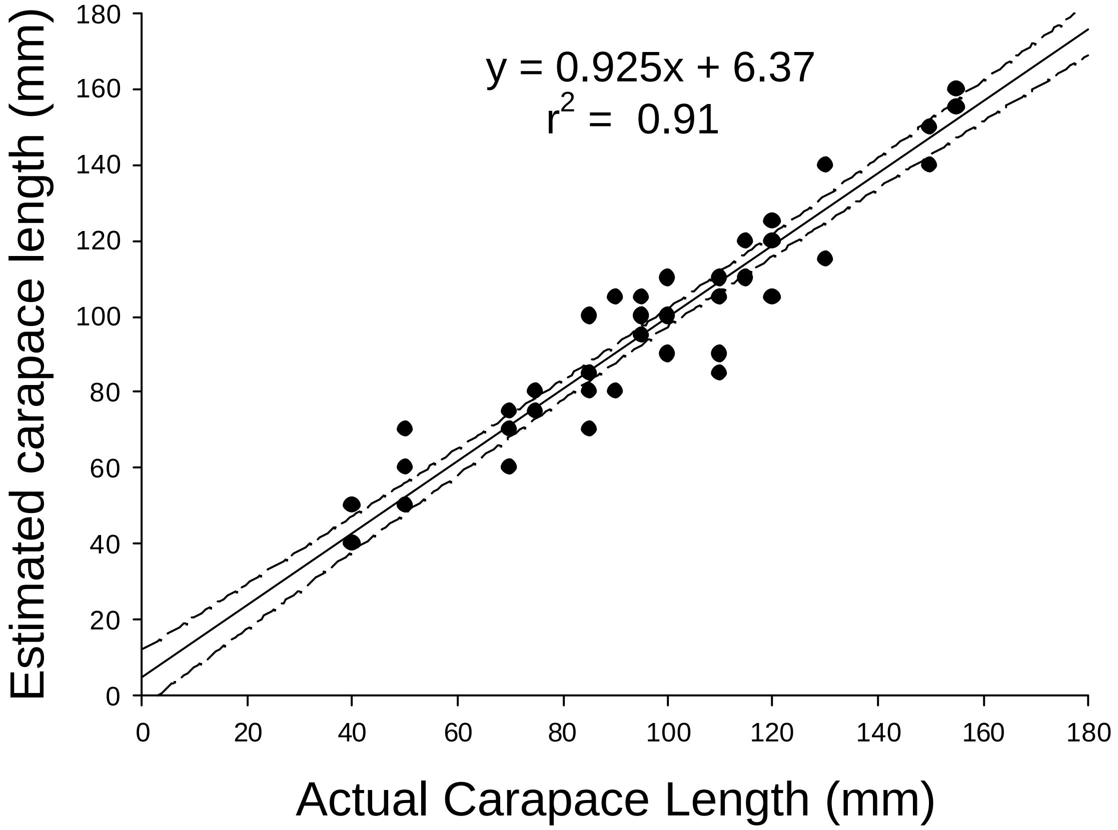

Figure 2. Diver visual size estimate calibrations for Crayfish surveys. Data are for three divers as indicated by symbols. The line indicates the slope of the regression and 95% confidence intervals.

2.5 Statistical analysis

Differences in abundance of crayfish and dominant fish species (BUV and UVC) between 2002 and 2003, and between inside and outside the marine park, were tested using a generalised linear mixed model with the GLMMIX procedure in SAS. These data are counts and therefore do not satisfy the assumptions of normality and homogeneity of variance required by ANOVA. Therefore, the model was fitted to a Poisson distribution (See Willis et al. (2000) for more details). The factors Survey (2002 and 2003) and Status (marine park and non-marine park) were treated as fixed factors and Site (Survey*Status) or Area (Survey*Status) was treated as a random effect. Ratios of density (plus 95% confidence limits) were calculated between levels of significant fixed factors to provide an estimate of the size of main effects. Note that confidence limits are asymmetrical as they are calculated on the log-scale.

Multivariate analyses were carried out using a Bray-Curtis similarity matrix calculated on square-root transformed abundance data for 32 species from UVC surveys. Similarities in reef fish assemblages among sites for each survey were investigated using principal coordinates analysis. The purpose of this analysis is to construct a 'map' of the samples in a specified number of dimensions, which attempts to satisfy all the conditions imposed by the rank similarity matrix. For example, if site 1 has a higher similarity to site 2 than it does to site 3 then it will be placed closer on the map to site 2 than it is to site 3. To determine whether there were any differences in overall fish community structure between surveys (2002 and 2003) or with park status (marine park and non-marine park areas), UVC data were analysed using non-parametric multivariate analysis of variance (NPMANOVA, Anderson 2001). A two-way crossed NPMANOVA using transect data pooled at the site level was carried out to test the overall effect of Survey and Status. A two-way nested NPMANOVA was then carried out on all transect data (not pooled at site level) to test the effect of Status and Site(Status) for each survey.

3. Results

3.1 Fish

3.1.1 Baited underwater video

Sites surveyed in 2003 were the same as those surveyed in 2002 and included similar numbers of sandy and rocky habitats in both areas, with slightly more gravel/sand habitats surveyed in the Marine Park. Sites surveyed in the Marine Park were slightly deeper on average (6-30 m depth range) than in the adjacent control areas (7-24 m depth range). These deeper sites were mainly in area 4 where the steeply sloping *Ecklonia radiata* covered reefs made it difficult to conduct shallower video drops.

Snapper

There was no significant difference among surveys in the relative density estimates for total or sublegal snapper, but there was for the density of legal sized snapper (Fig. 3, Table 1). Densities of legal snapper were 3.0 ($CL_{95\%}$=1.1, 8.0) times higher in 2003 (Table 1). There was no difference in total, legal or sublegal snapper between inside and outside the Marine Park (Fig. 3, Table 1) and this was consistent across both surveys (no significant survey*status interaction). While more legal snapper were recorded outside the marine park (n = 75) than inside (n = 62) in 2003 this was not significant ($F_{1,6}$=1.23, P=0.310).

Table 1. Results from mixed model analysis on BUV counts. Model back-fitted by removing non-significant interaction terms. Bold figures indicate significant p-values.

	Fixed effects			**Covariance Parameter Estimates**
	Survey	Status	Survey*Status	Area(Survey*Status)
Snapper				
Total	$F_{1,13}=0.01^{0.938}$	$F_{1,13}=0.37^{0.552}$	-	$0.01^{0.481}$
Legal	$F_{1,13}=4.77^{\mathbf{0.048}}$	$F_{1,13}=1.28^{0.278}$	-	0.00^{-}
Sublegal	$F_{1,13}=0.32^{0.581}$	$F_{1,13}=0.13^{0.723}$	-	0.00^{-}
Pigfish	$F_{1,13}=0.21^{0.655}$	$F_{1,13}=8.95^{\mathbf{0.010}}$	-	$0.05^{0.390}$
Leatherjacket	$F_{1,13}=4.53^{\mathbf{0.053}}$	$F_{1,13}=4.86^{\mathbf{0.046}}$	-	$0.40^{0.148}$

There was no significant difference in mean size between snapper inside and outside the park (Table 2). The mean size for snapper outside the Marine Park was slightly higher than for those inside in 2003, while in 2002 the opposite trend was observed. Overall the mean size of snapper measured in 2003 was 29 mm greater than in 2002.

Table 2. Average size (mm) of snapper per BUV (± s.e.) at all sites, the Marine Park (MP) and non Marine Park (NMP) at Mimiwhangata from autumn 2002 and 2003.

Snapper	2002 (autumn)	2003 (autumn)
All sites	204 (3.64)	233 (4.24)
MP	210 (4.58)	232 (6.56)
NMP	199 (5.83)	237 (6.24)

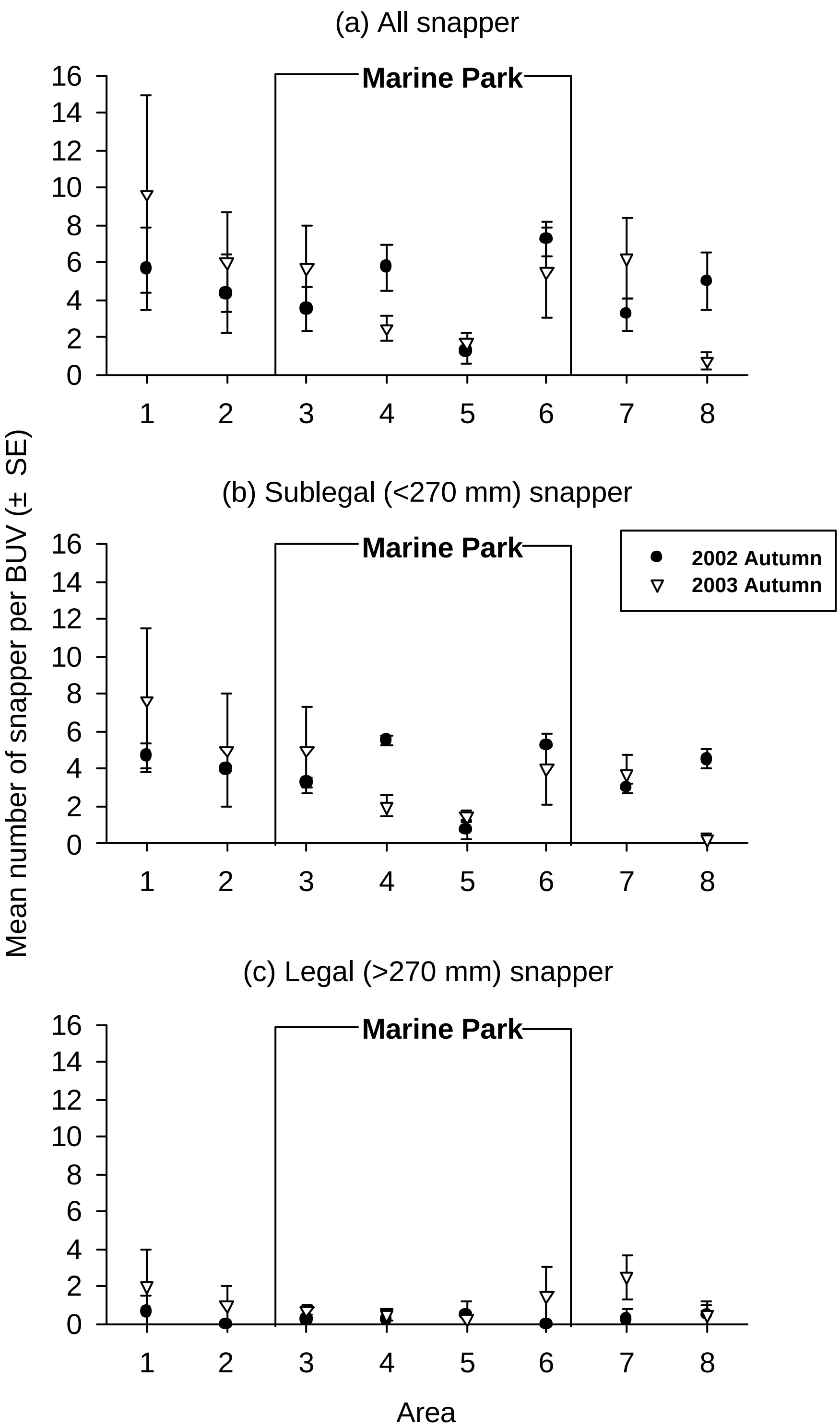

Figure. 3. Mean maximum number of (a) all snapper, (b) sublegal (<270mm) snapper and (c) legal (>270mm) snapper, *Pagrus auratus*, per baited underwater video (± s.e.) at 8 areas at Mimiwhangata, April 2003.

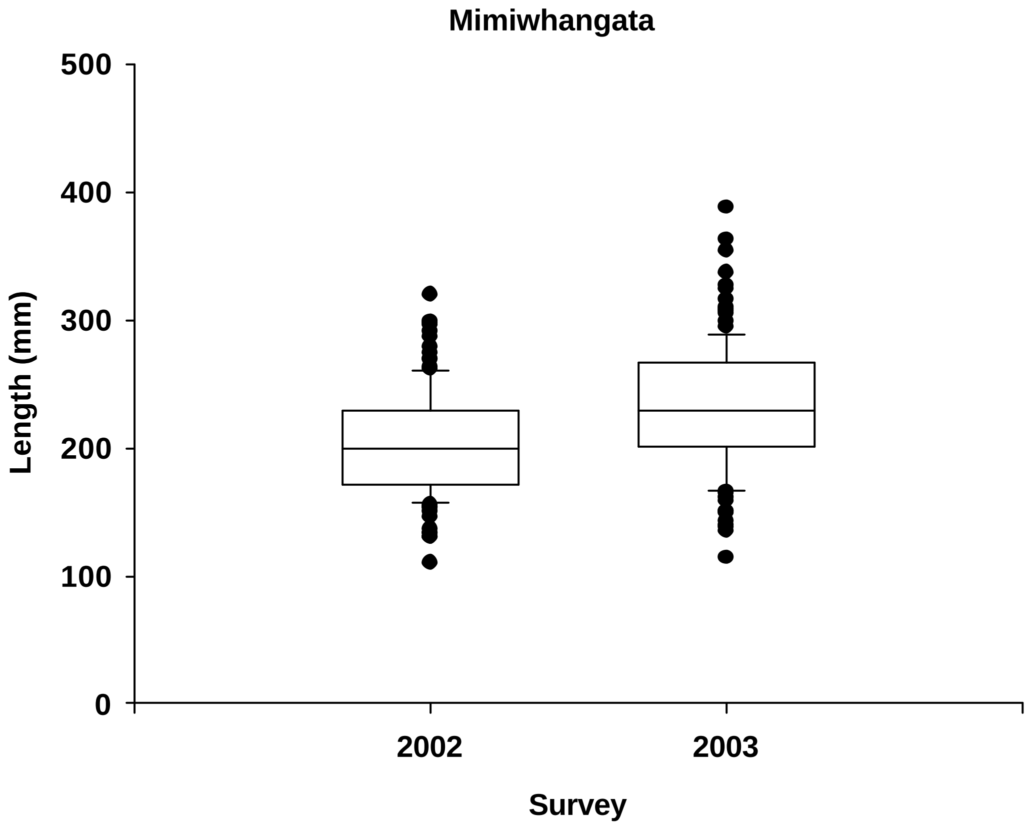

Figure 4. Box and whisker plot of snapper size at Mimiwhangata, autumn 2002 and 2003. The boundary of the box closest to zero indicated the 25th percentile, the line in the box represents the median, and the boundary of the box farthest from zero indicates the 75th percentile. The whiskers above and below the box indicate the 90th and 10th percentiles and the black circles represent outliers.

Other species

Other species recorded on the BUV were generally less abundant than snapper. Pigfish and Leatherjackets were commonly recorded and were found to be more abundant in the marine park (Table 1). Pigfish were 4.9 (1.7, 14.1) times more abundant in the park and Leatherjackets were 4.0 (1.2, 14.2) times more abundant in the park. Leatherjacket's also differed between the two surveys, being 3.7 (1.1,12.1) times more abundant in 2002. A number of schooling species such as Demoiselles, Sweep, Jack mackerel and Trevally were often recorded but these were highly variable and showed no difference between park and non-park areas.

3.1.2 Underwater visual census

The total number of fish species recorded by UVC in 2003 was 40 (Appendix 2). This included 9 species not recorded in 2002 (crimson cleanerfish, butterfly perch, painted moki, conger eel, yellow banded perch, blue cod, grey moray, long-tail and short-tail ray). Multivariate analyses were carried out on 32 species of reef fish. This did not include schooling species such as blue maomao, sweep, demoiselles, big eye, jack mackerel, kahawai and trevally as these are extremely variable and not reliably sampled using UVC.

Overall there were no clear differences in reef fish communities between years (Survey) or between the park and non-park areas (Status) (Table 3(a), Fig.5). The interaction between Survey and Status had a p-value of 0.08 suggesting a possible difference in the effect of status between years. Separate analyses for each year found that there was a difference between park and non-park areas in 2002 (Table 3(b)) but not in 2003 (Table 3(c)). The bi-plot for the principal coordinates ordination (Fig. 5) suggested that spotties, parore, leatherjackets, sandagers wrasse and pigfish were the species most responsible for the pattern of variation seen among sites. This pattern appears to reflect a gradient from more turbid areas (Areas 1 and 2), where more spotties and parore were recorded, to sites located on the headland (Areas 4 and 5), where more leatherjackets, pigfish and sandagers wrasse were recorded.

Table 3. Results from NPMANOVA comparing the reef fish assemblage between surveys and between park and non-park areas. Based on Bray-curtis similarities calculated on square-root transformed abundance data of 32 species.

Source	df	MS	F	P
(a) Both years				
Survey	1	1586.0	1.46	0.1584
Status	1	1622.5	1.49	0.1414
Survey*Status	1	1838.0	1.69	0.0860
(b) 2002				
Status	1	16005.9	2.68	**0.0282**
Site(Status)	14	5973.0	2.09	**0.0002**
(c) 2003				
Status	1	5581.1	0.99	0.4332
Site(Status)	14	5594.9	2.02	**0.0002**

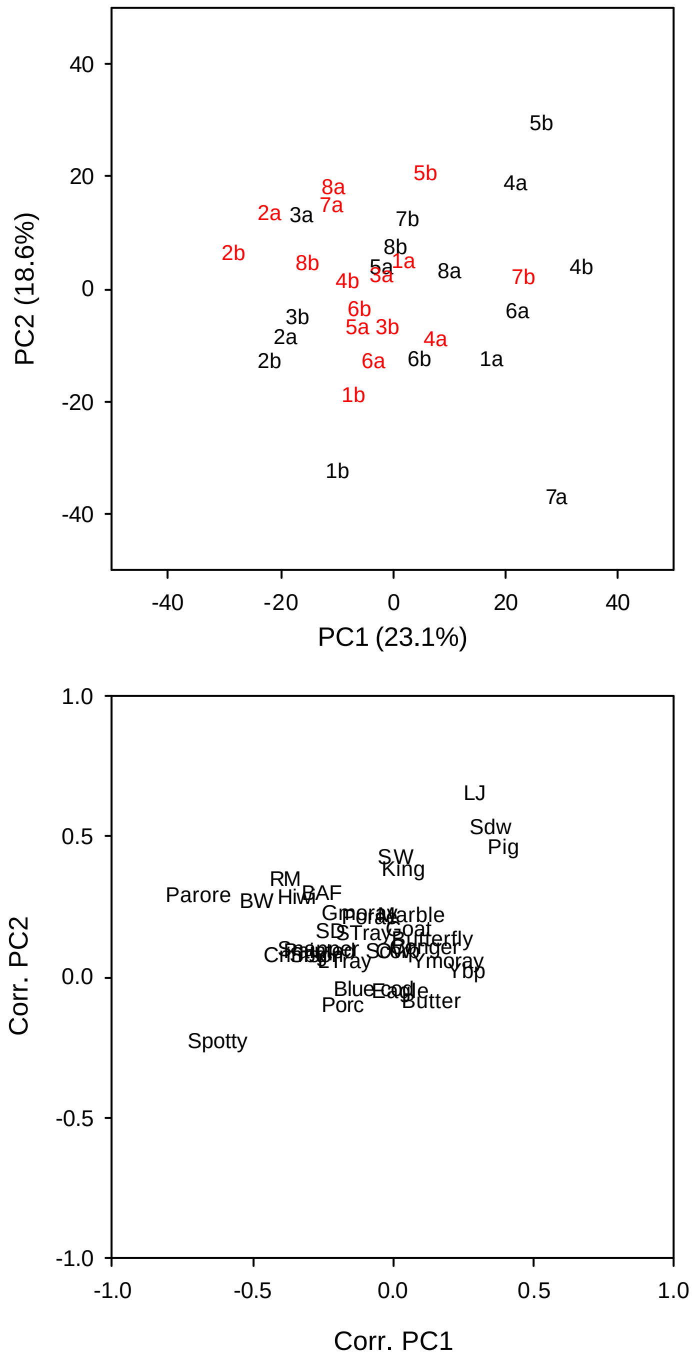

Figure 5. Principal coordinates analysis of the 16 sites based on square-root transformed UVC data for 32 species of reef fish at Mimiwhangata in April 2002 (black) and April 2003 (red). Sites are numbered according to the area in which they were located (the marine park includes areas 3-6). Note: The two sites in each area are identified as "a" and "b". The lower graph plots correlation coefficients of species abundances with principal coordinates axes 1 and 2.

Univariate comparisons of single species found a number of species to differ between 2002 and 2003, but few species showed any difference between inside and outside the marine park (Table 4). There was a large difference in the number of sweep recorded between the two sampling dates. Overall, sweep were 6.5 (2.1, 19.8) times more abundant in 2002. Higher numbers of blue maomao were also recorded in 2002 but this was not significant. The significant interaction between survey and status was explained by the higher abundance of blue maomao in the marine park in 2002 ($F_{1,14}$=4.33, P=0.056) but not in 2003 ($F_{1,14}$=3.91, P=0.068). A number of species were found to be more abundant in 2003, for example red moki, hiwihiwi and banded wrasse were 1.6 (1.1, 2.4), 2.3 (1.1, 4.9) and 2.8 (1.1, 7.0) times more abundant in 2003 respectively.

Goatfish were the only species that showed a consistent effect of status between years (Table 4, Fig. 7). On average goatfish were 2.7 (1.3, 5.9) times more abundant in the marine park. Leatherjackets also tended to be more abundant in the reserve in 2002 ($F_{1,14}$=3.86, P=0.069) but not in 2003 ($F_{1,14}$=1.11, P=0.311) as reflected by the significant interaction term (Table 4, Fig. 7). While pigfish showed a higher abundance in the marine park from BUV surveys only low numbers were recorded from UVC and there was no apparent effect of status (Table 4, Fig. 7).

Table 4. Univariate analyses (GLMMIX) for the 14 most common reef fish species recorded from UVC (bold values indicate a significant difference).

	Total numbers recorded		**Fixed effects**			**Covariance Parameter Estimates**
	2002	2003	Survey	Status	Survey*Status	Site(Survey*Status)
Sweep*	1398	289	$F_{1,29}=10.65^{\mathbf{0.003}}$	$F_{1,29}=0.05^{0.828}$	-	$1.08^{\mathbf{0.014}}$
Blue Maomao*	1029	653	$F_{1,28}=1.56^{0.222}$	$F_{1,28}=2.46^{0.128}$	$F_{1,28}=8.17^{\mathbf{0.008}}$	$1.12^{\mathbf{0.015}}$
Spotty	311	334	$F_{1,29}=0.10^{0.757}$	$F_{1,29}=1.23^{0.276}$	-	$0.44^{\mathbf{0.003}}$
Parore	185	313	$F_{1,29}=1.05^{0.314}$	$F_{1,29}=2.18^{0.151}$	-	$1.06^{\mathbf{0.006}}$
Goatfish	121	147	$F_{1,29}=0.64^{0.429}$	$F_{1,29}=6.47^{\mathbf{0.017}}$	-	$0.77^{\mathbf{0.002}}$
Red moki	99	168	$F_{1,29}=5.59^{\mathbf{0.025}}$	$F_{1,29}=0.00^{0.958}$	-	0.00^{-}
Leatherjacket	80	68	$F_{1,28}=0.10^{0.758}$	$F_{1,28}=0.50^{0.486}$	$F_{1,28}=4.80^{\mathbf{0.037}}$	$0.66^{\mathbf{0.014}}$
Black angelfish	22	19	$F_{1,29}=0.04^{0.845}$	$F_{1,29}=0.00^{0.978}$	-	$3.19^{\mathbf{0.004}}$
Sandagers	78	21	$F_{1,29}=0.91^{0.347}$	$F_{1,29}=0.39^{0.536}$	-	$2.97^{\mathbf{0.005}}$
Hiwihiwi	21	51	$F_{1,29}=4.97^{\mathbf{0.037}}$	$F_{1,29}=0.14^{0.714}$	-	$0.44^{\mathbf{0.054}}$
Banded wrasse	12	44	$F_{1,29}=5.06^{\mathbf{0.032}}$	$F_{1,29}=0.70^{0.410}$	-	$0.67^{\mathbf{0.021}}$
Pigfish	12	8	$F_{1,29}=0.16^{0.696}$	$F_{1,29}=0.61^{0.440}$	-	$1.68^{\mathbf{0.011}}$
Butterfish	6	10	$F_{1,29}=0.07^{0.801}$	$F_{1,29}=1.52^{0.227}$	-	$3.25^{\mathbf{0.006}}$
Snapper	3	14	$F_{1,29}=2.06^{0.162}$	$F_{1,29}=0.22^{0.643}$	-	$2.15^{\mathbf{0.013}}$

* Note: these two species were not included in multivariate analyses.

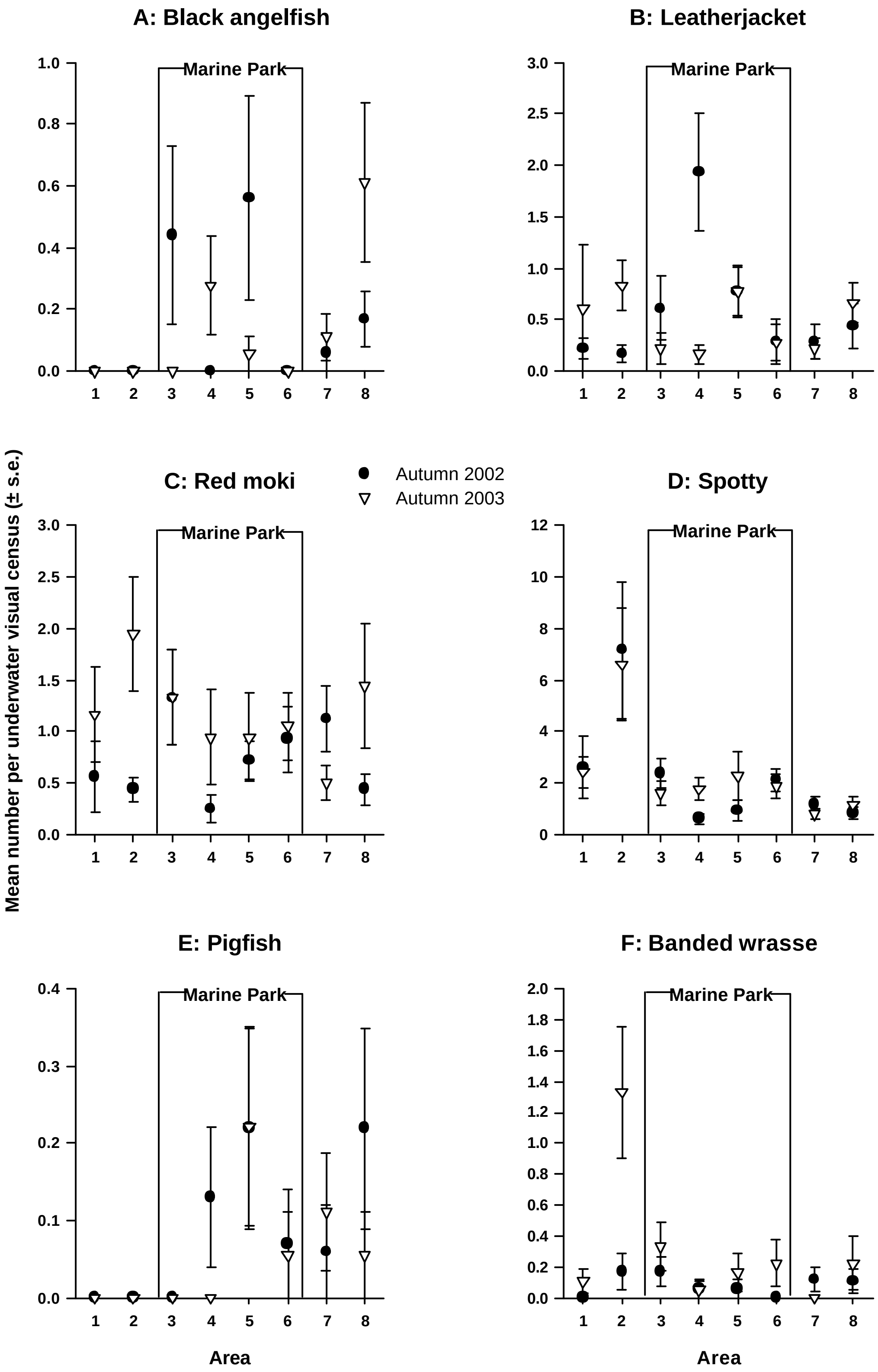

Fig. 6. Mean number of fish per underwater visual census (125m^2) (± s.e.) in 8 areas around Mimiwhangata; (A) *Parma alboscapularis*, black angelfish, (B) *Parika scaber*, leatherjacket, (C) *Cheilodactylus spectabilis*, red moki, (D) *Notolabrus celidotus*, spotty, (E) *Bodianus unimaculatus*, pigfish, and (F) *Notolabrus fucicola*, banded wrasse.

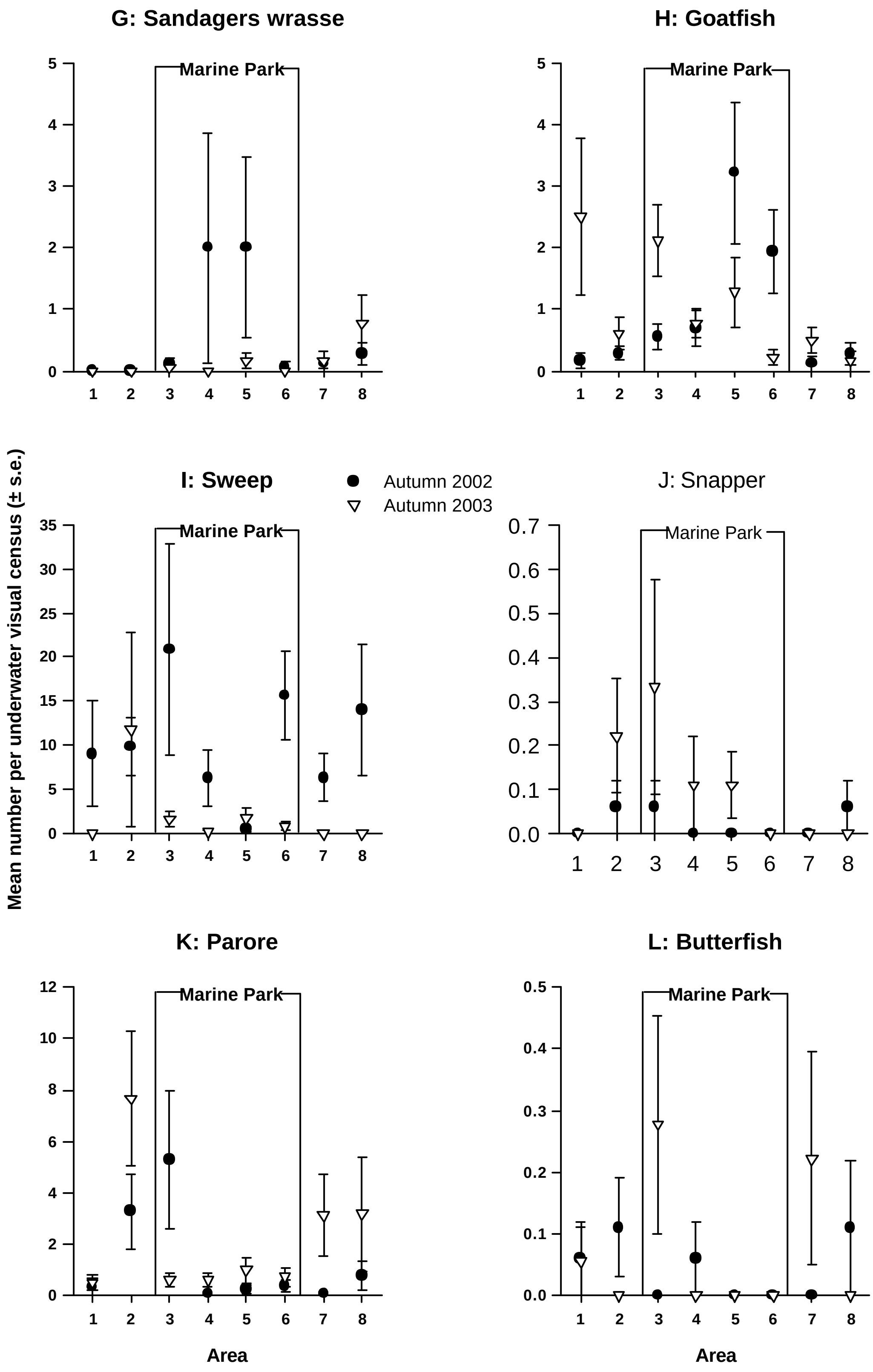

Fig. 7. Mean number of fish per underwater visual census ($125m^2$) (± s.e.) in 8 areas around Mimiwhangata; (G) *Coris sandageri*, sandagers wrasse, (H) *Upeneichthys lineatus*, goatfish, (I) *Scorpis lineolatus*, sweep, (J) *Chromis dispilus*, demoisielles, (K) *Girella tricuspidata*, parore, and (L) *Odax pullus*, butterfish.

3.2 Crayfish Underwater Visual Census

Overall densities of crayfish (*Jasus edwardsii*) were low, both inside the Marine Park and outside it (Fig. 8). Only 52 crayfish were recorded in 48 transects, 25 inside the Marine Park and 28 outside. There was no significant difference in density between the Marine Park and adjacent areas (Fig. 9, Table 6). While density did not vary significantly among areas nested within status, there was significant variation among sites. In general the highest numbers were recorded at some sites at the western end of the study area, regardless of whether they were inside the Marine Park. Nine Packhorse crayfish (*Jasus verreauxi*) were recorded during the study, and all were outside the marine park, seven in area 2 and two in area 7.

Table 6 Results from mixed model analysis on crayfish counts. Model back-fitted by removing non-significant interaction terms. Bold figures indicate significant p-values.

Fixed effects	**Covariance Parameter Estimates**
Status $F_{1,6}=0.00^{0.991}$	Area(Status) $0.89^{0.176}$
Status $F_{1,14}=0.04^{0.845}$	Site(Status) $1.52^{\mathbf{0.050}}$

Of the 27 crayfish for which sex was positively determined, 17 were female. This sex ratio was not significantly different from 1:1 ($?^2 = 1.12$, $p = 0.28$).

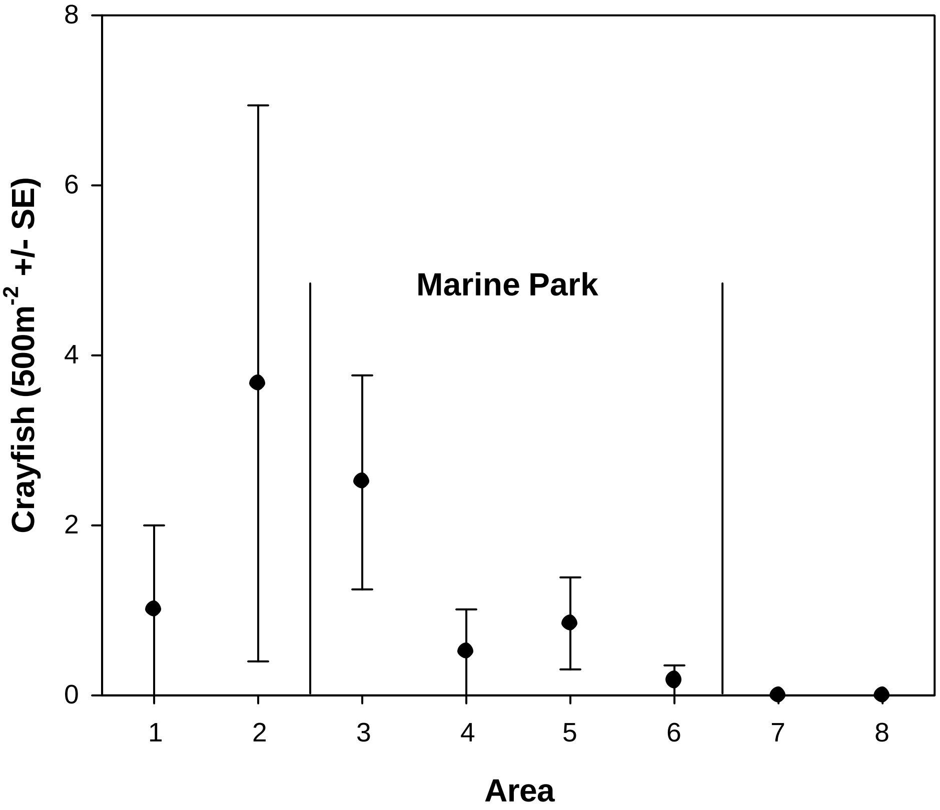

Figure 8. Crayfish density in survey areas, Mimiwhangata Marine Park 2003.

In general the large majority of crayfish measured in the survey were under the legal size-limit (95mm CL), with only 4 legal sized crayfish recorded at non-marine park sites and 5 inside the marine park (Fig. 9). There was no significant difference in the size of crayfish with respect to Marine Park status, with the mean size in the Marine Park 82.0 ? 6.4 mm CL and 87.5 ? 4.5 mm CL outside. In both marine park and non-marine park areas mean and modal size of crayfish were below the minimum legal limit of 95 mm carapace length (Fig. 10).

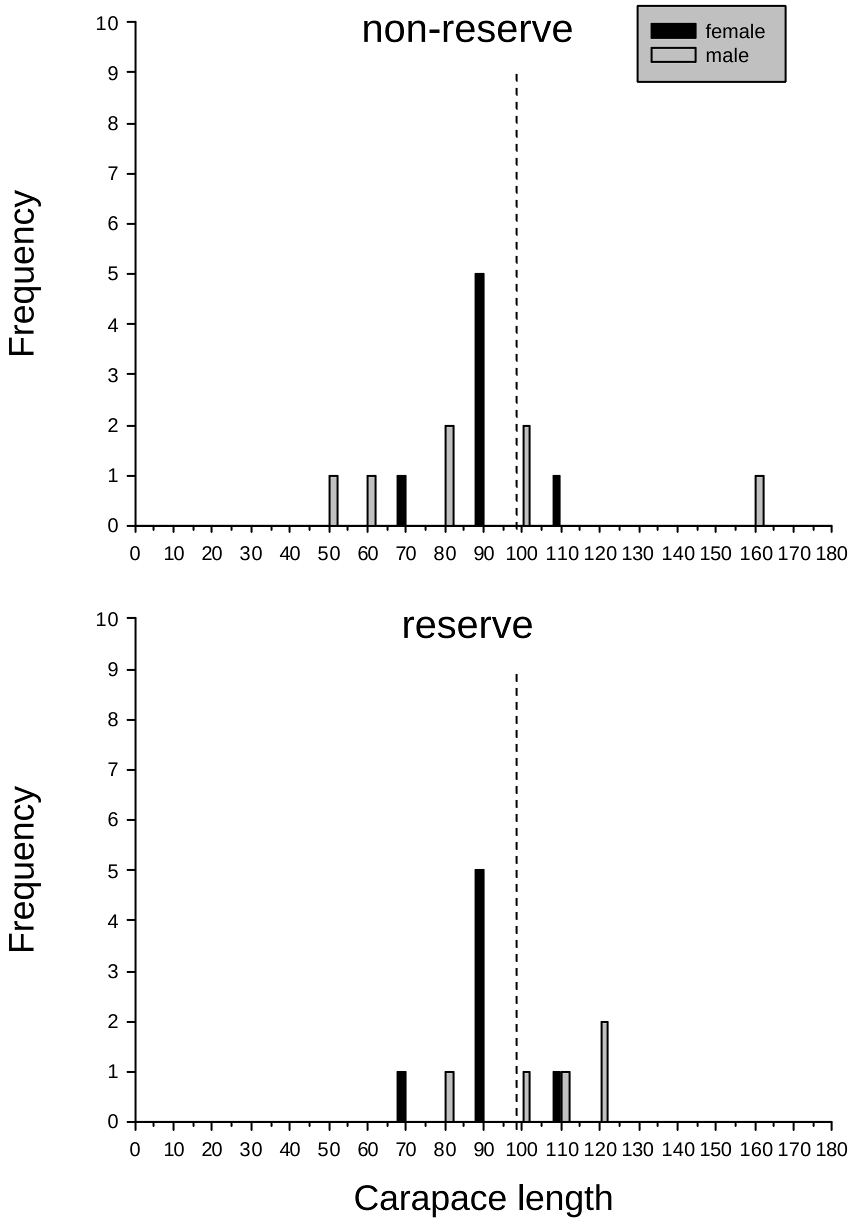

Figure 9. Crayfish population structure at Mimiwhangata, 2003. The histograms compare size frequencies of populations inside and outside the Mimiwhangata Marine Park. Dashed line indicates legal size limit of 95 mm carapace length.

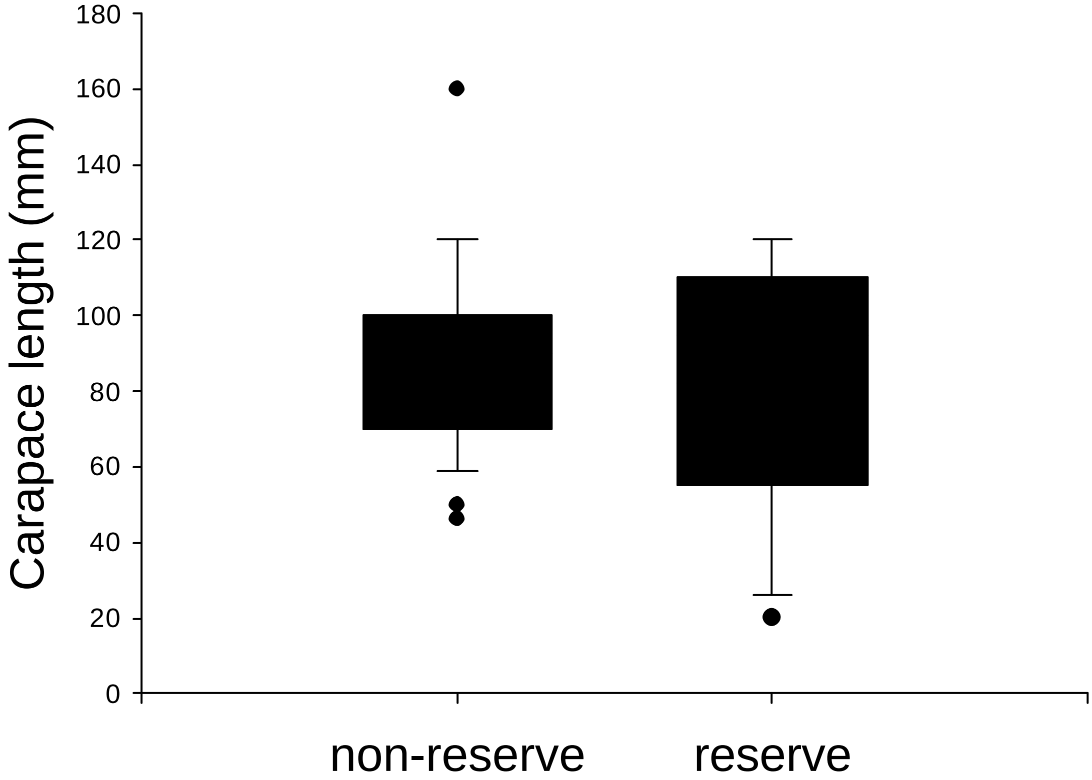

Fig. 10. Box and whisker plot of crayfish size at Mimiwhangata in 2003. The boundary of the box closest to zero indicated the 25th percentile, the solid line in the box represents the median, the dotted line the mean, and the boundary of the box farthest from zero indicates the 75th percentile. The whiskers above and below the box indicate the 90th and 10th percentiles and the black circles represent outliers.

4. Discussion

Snapper are the most heavily targeted recreational and commercial fish species throughout northeastern New Zealand, while crayfish are the most heavily targeted recreational and commercial invertebrate species. Where no-take marine reserves are in place, and enforced, the recovery of both these species has been dramatic, both in size and number (Kelly et al 2000, Denny and Babcock 2004, Willis et al 2003). Thus we should expect that if the gear and species restrictions at Mimiwhangata were in any way effective at protecting species such as snapper and crayfish then they would be more numerous and larger inside the Marine Park. However, when areas inside and outside the Marine Park were compared for snapper, there were almost identical numbers of snapper per baited underwater video and no significant difference in snapper size. Moreover, the number of legal-sized snapper tended to be lower at sites inside the marine park. Therefore, it appears that partial restrictions on gear and species are ineffective for this species. Restricting the use of weighted lines in the Marine Park is unlikely to protect snapper as, although taken on weighted lines, snapper can be caught effectively on unweighted lines, a practice permitted in the Marine Park. Similarly although no commercial crayfishing takes place in the park, recreational harvesting by divers, or by the single pots allowed, appears to be sufficient to restrict numbers to levels similar to those observed in areas able to be fished by commercial as well as recreational fishers.

Paradoxically, fishing pressure may even be higher within the Marine Park than outside it as there may be a perception that, in the absence of commercial fishing, fish are larger and more plentiful in Marine Parks. In addition, Marine Parks are often placed in areas that are pleasant to fish in, and consequently heavily utilised. Thus, Marine Park status and fishing gear restrictions at Mimiwhangata may, in fact, result in exactly the opposite pattern to the one intended. This possibility is supported by comparisons of snapper size and density at non-reserve sites in the region. Mimiwhangata had fewer and smaller snapper than two other unprotected areas in the region (Cape Brett or the Mokohinau Islands), probably due to high fishing pressure (Denny and Babcock 2004). This area is easily accessible to fishers from Tutukaka and from launching sites in Whangaruru/Oakura, and it is heavily fished during holiday periods (P. Bendle, pers. com., Denny and Babcock 2004). In France, Francour (1994) found that the density and biomass of fish on rocky reefs was lower in partially protected areas than unprotected areas. For example, the density of reef fish in a partially protected zone was 0.15 per 10 m^2 compared to 0.31 per 10 m^2 in an area with no protection.

Species that are targeted by spearfishers were seldom observed in visual transects transects. For example, in the 2002 survey no blue cod, three undersize snapper, and two porae were observed. In 2003 no blue cod were seen and only one porae. Although 14 snapper were seen only two were greater than legal size. This is in contrast to a pre-protection survey in 1973, in which it was noted that large snapper (“15-20 lbs”) were relatively common at Mimiwhangata (Ballantine et al., 1973). Spearfishing, a common activity at Mimiwhangata (P. Bendle, pers. com.) that tends to reinforce avoidance behaviour in fishes, may account for the low numbers of these species. Furthermore, the ability of spearfishers to selectively target large kingfish and snapper could lead to overall declines in the mean size and numbers of such species.

Based on the UVC data the overall fish assemblages within and outside the Marine Park did vary significantly with both year and status, however the species responsible for these differences were not consistent between the two surveys (Table 5). Only Spotties showed a consistent pattern in being significantly more abundant in areas outside the park in both 2002 and 2003. Other species have shown differences between the Marine Park and adjacent areas but have varied from year to year. This variability highlights the fact that fish abundance is inherently (and notoriously) variable and that multiple surveys are required in order to validly establish general patterns. The BUV data showed similar variability, with only Demoiselles showing a consistent pattern between years. This pattern is likely to be related to the topography and hydrography of the Marine Park area, which contains the Wide Berths Islands and is the most exposed part of coastline in the area. Such areas are likely to favour plankton feeding “offshore” species such as the demoiselle (Kingsford 1989). In contrast, the more sheltered bays to the west of the Marine Park may provide more appropriate habitat for other fish species such as spotty, *Notolabrus celidotus*, which tended to be more abundant in these areas (Area 1 and 2). The differences in fish communities identified between areas inside and outside the marine park are therefore most likely due to the large variation in environmental conditions around the headland where the park is situated. Similar habitat related factors are likely to influence the current distribution of crayfish throughout the area and may explain the greater abundance’s recorded in areas west of the marine park.

This study demonstrates that the partial closures at Mimiwhangata are ineffective as conservation tools either for heavily targeted species such as snapper or crayfish, or for fish communities in general (i.e. through reduction in by-catch). The fact that snapper numbers may actually be lower in the partially protected Marine Park than in the unprotected control areas begs the question; is no protection at all better than partial protection? This may be so for two reasons: firstly, partial reserves may give a false impression that a conservation outcome has been achieved. Secondly, this impression may focus fishing effort, locally resulting in even greater fishing effects. The findings of this study have important implications for conservation managers, many of whom have had to accept the provision of fishing within a marine reserve as a 'solution' to political issues surrounding the declaration of marine reserves. This was because there was a lack of evidence either for or against the effects of limited fishing within a marine reserve. In light of the results in this study, we conclude that only no-take marine reserves should be created, as partial protection is an ineffective conservation strategy.

5. Acknowledgements

We wish to thank Tony W, Charlie B, James Williams, Caroline Williams and Phil Ross for their help diving. Also skipper Phil Bendle for his enthusiasm when carrying out these surveys. In addition, thanks to Roger Grace for information about his sampling techniques. Finally thanks to Vince Kerr and the Northland Department of Conservation for their foresight in funding this work.

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7. Appendix 1. List of fish and invertebrate species allowed to be taken by recreational fishers in the Mimiwhangata Marin Park.

<u>Permitted list</u>:

Fin fish	Barracouta
	Mackerel (all types)
	Billfish (all types)
	Piper (garfish)
	Blue maomao
	Shark (all types)
	Flounder (all types)
	Snapper
	Grey mullet
	Sole
	Yellow eye mullet
	Tarakihi
	Gurnard
	Trevally
	Kahawai
	Tuna (all types)
	Kingfish
Shellfish	Common kina
	Scallop
	Green-lipped mussel
	Tuatua
	Rock lobster

Other species:	All other species of finfish, shellfish, and other marine life are totally protected.

8. Appendix 2. Alphabetical species list for fish counted by UVC and BUV at Mimiwhangata in 2003.

Acanthistius cinctus	Yellow banded perch
Allomycterus jaculiferus	pufferfish
Aplodactylus arctidens	Marblefish
Arripis trutta	Kahawai
Bodianus unimaculatus	pigfish
Caesioperca lepidoptera	Butterfly perch
Centrobeyx affinis	Golden snapper
Cheilodactylus ephippium	Painted moki
Cheilodactylus spectabilis	Red moki
Chironemus marmoratus	hiwihiwi
Chromis dispilus	Demoiselle
Conger verreauxi	Conger eel
Coris sandageri	Sandagers wrasse
Dasyatis brevicaudata	Short-tailed stingray
Dasyatis thetidis	Long-tailed stingray
Decapterus koheru	Koheru
Epinephelus daemelii	Spotted black grouper
Girella tricuspidata	parore
Gymnothorax nubilis	Grey moray
Gymnothorax prasinus	Yellow moray
Hypoplectrodes sp.	Half banded perch
Kyphosus sydneyanus	Drummer
Myliobatus tenuicaudatus	Eagle ray
Nemadactylus douglasii	Porae
Notolabrus fucicola	Banded wrasse
Obliquichthys maryannae	Oblique swimming triplefin
Odax pullus	Butterfish
Optivus elongatus	Slender roughy
Pagrus auratus	Snapper
Parapercis colias	Blue cod
Parika scaber	Leatherjacket
Parma alboscapularis	Black angelfish
Pempheris adspersus	Bigeye
Pseudocaranx dentex	Trevally
Pseudolabrus miles	Scarlet wrasse
Scorpis lineolatus	Sweep

Scorpis violaceus	Blue maomao
Seriola lalandi	Kingfish
Suezichthys aylingi	Crimson cleanerfish
Trachurus novaezelandiae	Jack mackerel
Upeneichthys lineatus	Goatfish
Zeus faber	John dory