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Do partial marine reserves protect reef fish assemblages?

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Abstract

Fish assemblages in the Mimiwhangata Marine Park, an area closed to commercial fishing but open to most forms of recreational fishing, were compared with adjacent fished areas. 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. When compared to the fully no-take Poor Knights Island Marine Reserve and two other reference areas open to all kinds of fishing (Cape Brett and the Mokohinau Islands), the abundance and size of snapper at the Marine Park were most similar to fished reference areas. In fact, the Marine Park had the lowest mean numbers and sizes of snapper of all areas, no-take or open to fishing. Baited underwater video found that pigfish (*Bodianus unimaculatus*), leatherjackets (*Parika scaber*) and trevally (*Pseudocaranx dentex*) were significantly more common in the Marine Park, than in the adjacent control areas. However, none of these species are heavily targeted by fishers. Underwater visual census found similar results with five species significantly more abundant in the Marine Park and five species more abundant outside the Marine Park. The lack of any recovery by snapper within the Marine Park, despite the exclusion of commercial fishers and restrictions on recreational fishing, indicates that partial closures are ineffective as conservation tools. The data suggest fishing pressure within the Marine Park is at least as high as at other 'fished' sites.

Keywords: Marine protected area; Mimiwhangata Marine Park; *Pagrus auratus*; Snapper; New Zealand; Gear restrictions

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 Meditteranean 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).

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

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rates. Special fisheries regulations exist at Mimiwhangata prohibiting all commercial fishing, nets and long-lines. However, recreational fishers may use the following methods: unweighted, single-hook lines, trolling and spearing. A number of species are permitted to be caught within the Marine Park, all thought to be nomadic or pelagic at the time of the park's creation (see Table 3 for takeable species). That is, they were not considered part of the resident demersal reef fish assemblage. However, 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 wholly or partially resident of 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.

The main objective of this survey was to evaluate the effectiveness of partial protection on the reef fish assemblages within and around the Mimiwhangata Marine Park. Furthermore, snapper abundance at Mimiwhangata was compared with data from three nearby areas in northern New Zealand, the no-take Poor Knights Islands Marine Reserve, the Mokohinau Islands and Cape Brett which are both fully open to fishing (Fig. 1). In this survey, two different methods were used to provide quantitative estimates of fish abundance and size; underwater visual census and baited underwater video.

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 four areas, and these were compared with four control areas outside the Marine Park (two at either end of the Marine Park) to assess differences inside and outside the Marine Park (Fig. 2). 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, 2003). This design has the dual advantages of ensuring reference areas are similar to reserve areas, as well as enabling the detection

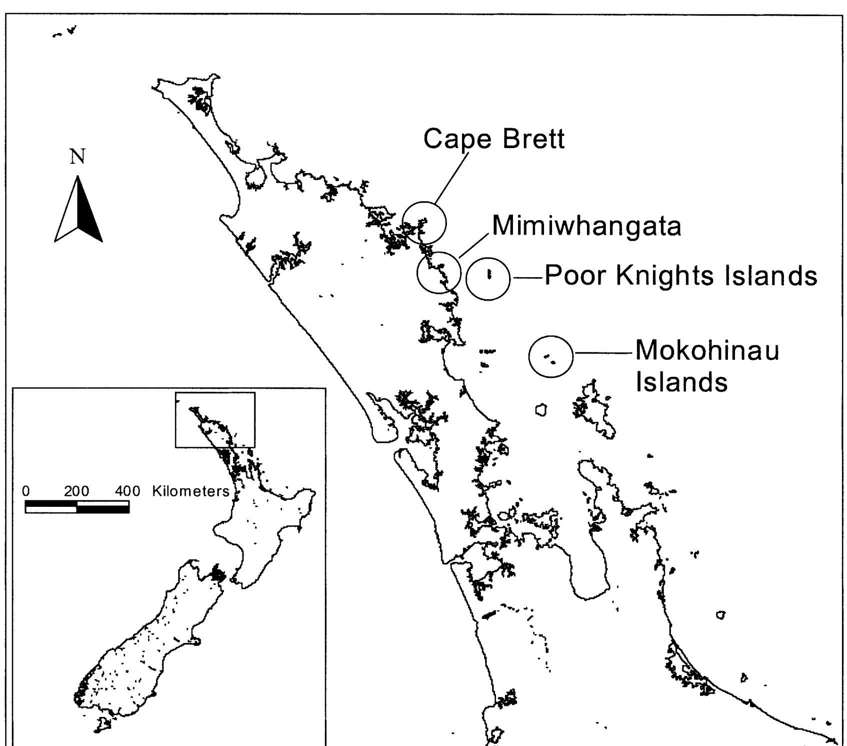

Fig. 1. Map of northern New Zealand showing the location of Mimiwhangata, Cape Brett, the Poor Knights and Mokohinau Islands.

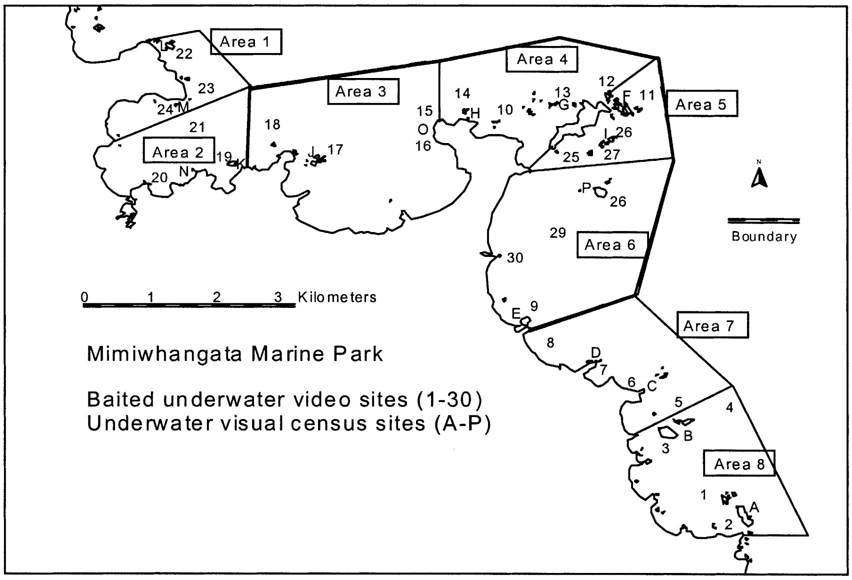

Fig. 2. Map of Mimiwhangata showing the location of the baited underwater video sites (1–30) and the underwater visual census sites (A–P) in April 2002.

of any edge effects that might be related to the encroachment of fishing effects into the reserve, (or alternatively spillover). Sampling was conducted between 08:00 and 17:00 h from 2 to 5 April 2002. Data were collected concurrently at three additional locations in northeastern New Zealand as part of a related study; two island locations (the Poor Knights Islands and Mokohinau Islands) and another mainland location (Cape Brett) (Fig. 1). Their inclusion in this study provided an example of how snapper numbers in other fished and unfished areas of northern New Zealand compare with Mimiwhangata. Biogeographic differences between these sites and Mimiwhangata limit their usefulness in the context of comparing whole fish assemblages (Choat and Ayling, 1987; Brook, 2002).

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 m above a bait container holding approximately 300 g 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 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. 2), except areas one and two where three replicate drops were done (due to logistical constraints). Thirty replicate drops were also conducted at the Poor Knights, Cape Brett, and the Mokohinau Islands (for locations see Willis and Denny, 2000). A total of 60 h of videotape was collected for later analysis.

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 (30 counts made during each 30-min 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 more conservative abundance estimates in high density areas than low density areas, and therefore observed relative differences between sites are also likely to be conservative.

2.3. 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 (<30 m), 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, two sites within each of the eight areas at Mimiwhangata were surveyed by underwater visual census (16 sites in total; Fig. 2). At least 16 sites were surveyed by underwater visual census at each of the three other locations. Three divers recorded the numbers of all fish and the size of several selected species vulnerable to fishing using 5 m×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 (9×125 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 m to avoid counting species attracted to the initial activity, and preceded to swim 25 m, counting all fish within a strip 2.5 m either side of the diver (Denny et al., 2003). All divers had previous experience using this methodology.

2.4. Statistical analysis

The baited underwater video data are counts and therefore do not satisfy the assumptions of normality and homogeneity of variance that are required by ANOVA. Therefore, the video data were analysed using the Poisson distribution using the GENMOD procedure in SAS to obtain unbiased estimates of relative abundance for dominant carnivorous species. See Willis et al. (2000) for a more detailed description of this analysis.

To determine whether there were any differences in overall fish community structure between fished and unfished areas, underwater visual census data were analysed using metric multidimensional scaling in the CAP statistical package (Anderson, 2002). Site transect data were pooled, square root transformed, and a Bray–Curtis similarity matrix was generated. The purpose of multidimensional scaling is to construct a ‘map’ of configuration 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 site 1 will be placed closer on the map to site 2 than it is to site 3. For single species, comparisons were made using the GENMOD procedure in SAS, as described for the video analysis.

3. Results

3.1. Baited underwater video

Similar numbers of sandy and rocky habitats were surveyed in both areas and slightly more gravel/sand habitats were 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 four where the steeply sloping *Ecklonia radiata* covered reefs made it difficult to conduct shallower video drops.

There was no significant difference between the mean maximum number of snapper per baited underwater video inside (4.44±1.15 S.E.) and outside the Marine Park (4.5±1.59). Numbers of sublegal snapper (fish too small to be legally taken, <270 mm SL) mirrored the pattern of all snapper (Fig. 3a), as these smaller fish made up the bulk of snapper recorded (Fig. 3b). There were very low numbers of legal snapper (those fish that can be legally retained, >270 mm SL) in any area (Fig. 3c). Comparisons with the other locations showed that Mimiwhangata had the lowest mean snapper numbers, particularly legal size snapper (Table 1). Interestingly, the mean number of sublegal (<270 mm) snapper at Mimiwhangata was similar to sublegal snapper numbers at the Poor Knights and the Mokohinau Islands (Table 1).

Out of the 126 snapper measured at Mimiwhangata, 117 were under the legal minimum size of 270 mm. The average snapper size inside the Marine Park was 209 mm (±4.6), slightly larger than in the control area at 199 mm (±5.8), however, this difference was not sig-

Table 1
Mean number of all, legal (>270 mm) and sublegal (<270 mm) snapper per baited underwater video (±S.E. in parentheses) at the Poor Knights, Cape Brett, Mokohinau Islands, and Mimiwhangata in autumn 2002

Snapper	Autumn 2002 (April/May)			
	Poor Knights	Cape Brett	Mokohinau Is.	Mimiwhangata
All	16.9 (2)	11.5 (1.2)	5.6 (0.8)	4.13 (0.9)
Legal (>270 mm)	11.5 (1.2)	1.5 (0.4)	0.9 (0.3)	0.3 (0.1)
Sublegal (<270 mm)	4.4 (0.9)	9.75 (1.2)	4.8 (0.8)	3.83 (0.9)

nificant ($P=0.7$). Overall, the average snapper size at Mimiwhangata was 204 mm (±3.6), significantly lower ($P<0.01$) than at Cape Brett (221 mm±2.3) and the Mokohinau Islands 227 mm (±3.4; Fig. 4). The average size at the no-take Poor Knights Islands Marine Reserve was 310 mm (±3.2; Fig. 4). Large fish (>350 mm), recorded at other areas, were not seen at Mimiwhangata where the largest snapper was only 320 mm (Fig. 4).

Analysis of the baited underwater video data found that of the seven most commonly recorded species, pigfish (*Bodianus unimaculatus*), leatherjackets (*Parika scaber*) demoiselles (*Chromis dispilus*) and trevally were significantly more common in the Marine Park than in adjacent areas (Table 2). Only sweep (*Scorpis lineolatus*) were significantly more common in the control areas (Table 2).

3.2. Underwater visual census

Species richness at Mimiwhangata (31 species) was much lower than at the other three survey areas, where 40 species were recorded at Cape Brett, 49 at the Poor Knights, and 43 at the Mokohinau Islands. Species at Mimiwhangata were characteristic of the mainland species observed at Cape Brett and only a few of the subtropical species found at the Poor Knights and Mokohinau Islands were recorded there.

Densities of the 12 most common fish species recorded at Mimiwhangata were highly variable both within and between sites (Figs. 5 and 6). There was little differentiation in fish communities between Marine Park sites and control sites (Fig. 7). The majority of species

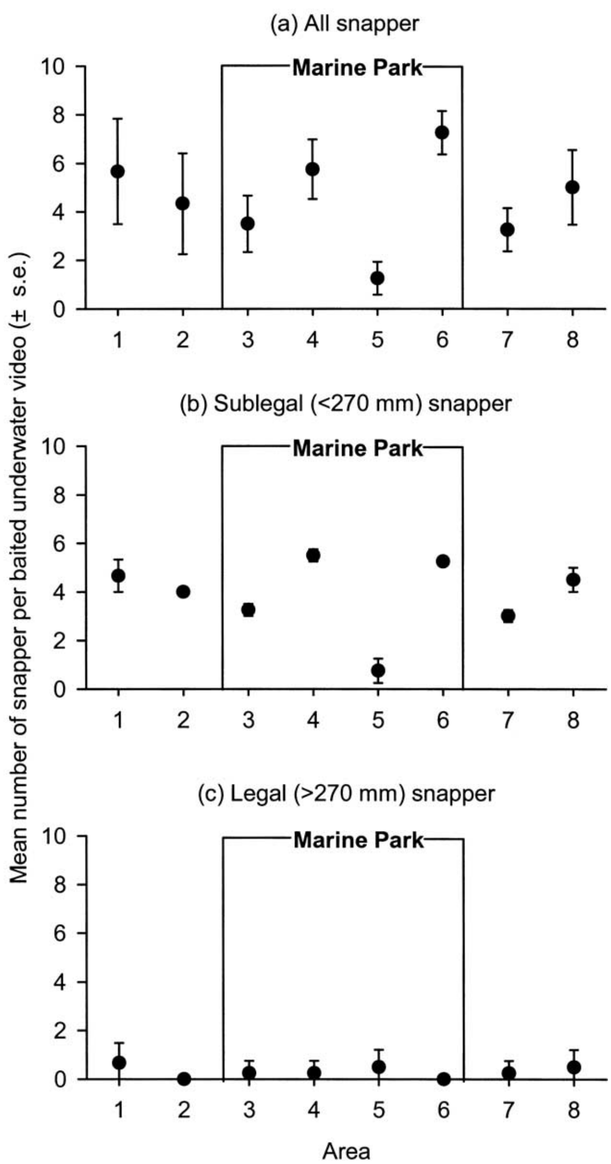

Fig. 3. Mean maximum number of (a) all snapper, (b) sublegal (<270 mm) snapper and (c) legal (>270 mm) snapper per baited underwater video (±S.E.) at eight areas at Mimiwhangata.

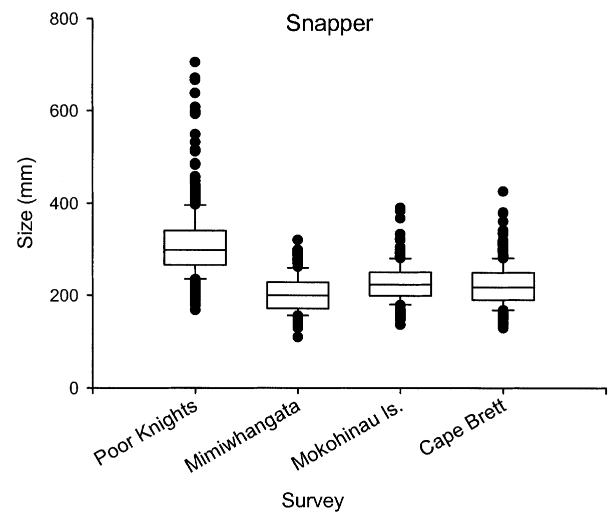

Fig. 4. Box and whisker plot of snapper at Mimiwhangata, Mokohinau Islands, Cape Brett, and the Poor Knights Islands, autumn 2002. The boundary of the box closest to zero indicated the 25th percentile, the line in the box represents the median, and the boundary of the box farthest from zero indicates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

Table 2
Differences in fish density of seven species inside and outside the Mimiwhangata Marine Park, estimated by baited underwater video in April 2002 with 95% lower and upper confidence limits (CL)

Species	Abundance ratio	95% Lower CL	95% Upper CL	χ^2	*P*-value
Snapper	1.01	0.72	1.42	0.01	0.94
Pigfish	3.06	1.0	9.3	3.9	0.048*
Leatherjacket	4.96	2.08	11.8	13.07	<0.01*
Demoiselles	10.5	1.37	80.8	5.1	0.024*
Trevally	10.9	2.59	46.2	10.6	0.01*
Sweep	0.55	1.3	2.7	12.33	0.01*
Yellow moray	1.31	0.22	7.9	0.09	0.77

* Indicates species whose abundance ratio is significantly different at the $P<0.01$ value.

censused (21/31) showed no significant difference in density between the Marine Park and adjacent control sites. However, in spite of there being no significant difference in overall fish assemblages, some species were significantly more common in the Marine Park. These were black angelfish (*Parma alboscapularis*), leatherjackets, sandagers wrasse (*Coris sandageri*), goatfish (*Upeneichthys lineatus*) and blue maomao (*Scorpis violaceus*; Table 3). Conversely, other species were significantly more common outside the Marine Park. These were spotties (*Notolabrus celidotus*), demoiselles, sweep, jack mackerel (*Trachurus novaezelandiae*) and koheru (*Decapterus koheru*; Table 3). Interestingly, in areas where pigfish were absent (areas 1–3), spotties occurred in high numbers (Fig. 5D and E). Other species such as orange wrasse (*Pseudolabrus luculentus*) and scarlet wrasse (*P. miles*) common at the Poor Knights and Cape Brett were rare at Mimiwhangata.

4. Discussion

Snapper is the most heavily targeted recreational and commercial fish species throughout northeastern New Zealand. Where no-take marine reserves are in place, and enforced, the recovery of this species has been

Table 3
Scientific name, species, family, abundance ratio, 95% lower and upper confidence limits, χ^2 and *P*-values of 31 fish species observed in underwater visual census at the Mimiwhangata Marine Park in April 2002

Scientific name	Species	Family	Abundance ratio	95% Lower CL	95% Upper CL	χ^2	*P*-value
Allomycterus jaculiferus	Porcupinefish	Diodontidae	No fit				
Aplodactylus arctidens	Marblefish	Aplodactylidae	0.5	0.09	2.7	0.64	0.42
Arripis trutta	Kahawai[a]	Arripadae	1.36	0.79	2.36	1.22	0.29
Bodianus unimaculatus	Pigfish[b]	Labridae	1.4	0.44	4.41	37.83	0.56
Cheilodactylus spectabilis	Red moki	Cheilodactylidae	0.98	0.66	1.45	0.01	0.92
Chironemus marmoratus	Hiwihiwi[b]	Chironemidae	1.09	0.47	2.59	0.05	0.83
Chromis dispilus	Demoiselle	Pomacentridae	0.76	0.70	0.83	36.8	<0.01*
Coris sandageri	Sandagers wrasse[b]	Labridae	10.1	4.67	22.1	34.2	<0.01*
Decapterus koheru	Koheru[a]	Carangidae	0.47	0.38	0.58	47.32	<0.01*
Epinephelus daemelii	Spotted black grouper[b]	Serranidae	No fit				
Girella tricuspidate	Parore	Girellidae	1.18	0.88	1.57	1.21	0.27
Gymnothorax prasinus	Yellow moray[b]	Muraenidae	No fit				
Kyphosus sydneyanus	Silver drummer	Kyphosidae	No fit				
Myliobatus tenuicaudatus	Eagle ray	Myliobatidae	No fit				
Nemadactylus douglasii	Porae[b]	Cheilodactylidae	No fit				
Notolabrus celidotus	Spotty[b]	Labridae	0.47	0.37	0.47	37.83	<0.01*
Notolabrus fucicola	Banded wrasse[b]	Labridae	0.71	0.23	2.25	0.33	0.57
Obliquichthys maryannae	Oblique swimming triplefins	Tripterygiidae	No fit				
Odax pullus	Butterfish	Odacidae	0.2	0.02	1.71	2.16	0.14
Pagrus auratus	Snapper[a]	Sparidae	0.5	0.05	5.5	0.32	0.57
Parika scaber	Leatherjacket	Monacanthidae	2.99	1.81	4.98	18.1	<0.01*
Parma alboscapularis	Black angelfish	Pomacentridae	4.5	1.52	13.3	7.4	<0.01*
Pempheris adspersus	Bigeye	Pempheridae	No fit				
Pseudolabrus luculentus	Orange wrasse	Labridae	No fit				
Pseudolabrus miles	Scarlet wrasse[b]	Labridae	1	0.14	7.1	0	1
Scorpaena cardinalis	Northern scorpionfish[b]	Scorpaenidae	No fit				
Scorpis lineolatus	Sweep[b]	Scorpidae	0.59	0.53	0.66	88.63	<0.01*
Scorpis violaceus	Blue maomao[b]	Scorpidae	47.9	31.2	73.9	308.3	<0.01*
Seriola lalandi	Kingfish[a]	Carangidae	No fit				
Trachurus novaezelandiae	Jack mackerel[a]	Carangidae	0.33	0.29	0.37	384.6	<0.01*
Upeneichthys lineatus	Goatfish	Mullidae	4.5	2.84	7.14	40.72	<0.01*
Takeable species not observed in this study							
Thyrsites atun	Barracouta[a]	Gempylidae					
Pseudocaranx dentex	Trevally[a]	Carangidae					
	Tuna—6 species[a]	Scombridae					
	Billfishes—6 species[a]	Istiophoridae					
	Mackerel—5 species[a]	Carangidae					
	Sharks—27 species[a]	Many families					

[a] Signifies species permitted to be caught.
[b] Signifies species known to be caught as by-catch.
* Indicates species whose abundance ratio is significantly different at the $P<0.01$ value.

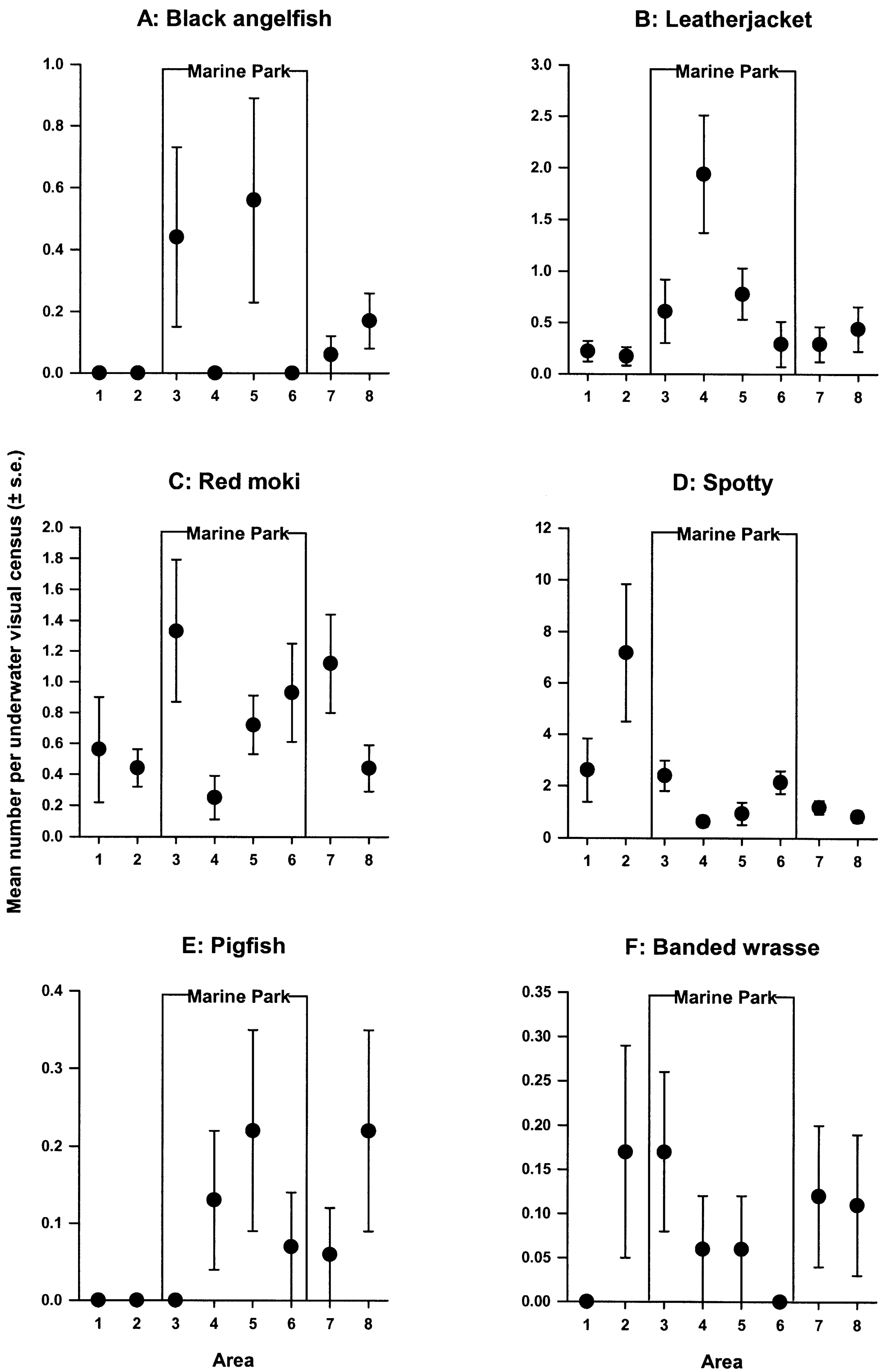

Fig. 5. Mean number of fish per underwater visual census (125 m^2) (±S.E.) in eight areas around Mimiwhangata; (A) *Parma alboscapularis*, black angelfish, (B) *Parika scaber*, leatherjacket, (C) *Cheilodactylus spectabilis*, red moki, (D) *Notolabrus celidotus*, spotty, (E) *Bodianus unimaculatus*, pigfish, and (F) *Notolabrus fucicola*, banded wrasse.

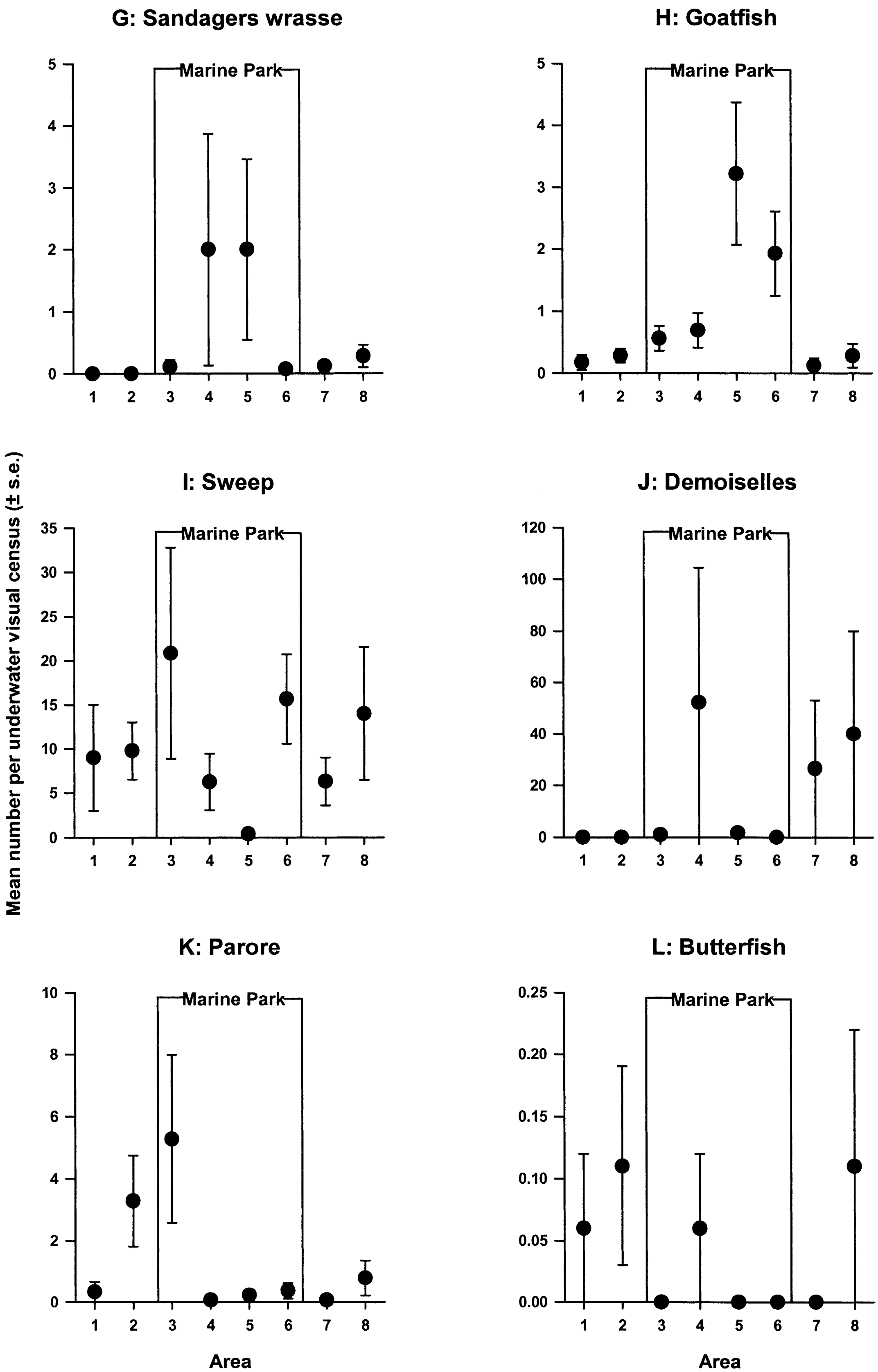

Fig. 6. Mean number of fish per underwater visual census (125 m^2) (±S.E.) in eight areas around Mimiwhangata; (G) *Coris sandageri*, sandagers wrasse, (H) *Upeneichthys lineatus*, goatfish, (I) *Scorpis lineolatus*, sweep, (J) *Chromis dispilus*, demoiselles, (K) *Girella tricuspidata*, parore, and (L) *Odax pullus*, butterfish.

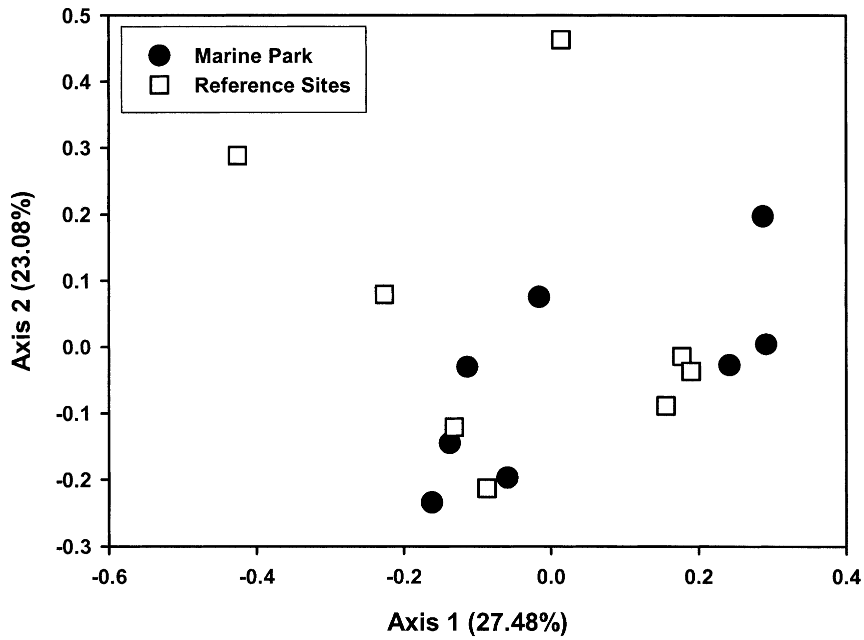

Fig. 7. Metric multidimensional scaling ordination plot of the 16 sites (pooled replicates) based on underwater visual census surveys of 28 species of reef fish at Mimiwhangata in April 2002.

dramatic, both in size and number (Table 4). Thus we should expect that if the gear and species restrictions at Mimiwhangata were in any way effective at protecting snapper, there would be more numerous and larger snapper inside the Marine Park. However, when areas inside and outside the Marine Park were compared, there were almost identical numbers of snapper numbers per baited underwater video and no significant difference in snapper size. 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.

Mimiwhangata had fewer and smaller snapper than either of the unprotected areas, Cape Brett or the Mokohinau Islands, probably due to high fishing pressure. 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, personal communication).

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. 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 with 0.31 per 10 m^2 in an area with no protection.

Table 4
Northeastern New Zealand sites surveyed with baited underwater video to assess relative legal sized (>270 mm) snapper abundance and the reserve: non-reserve snapper ratio

Location	Year	Reserve mean	Non-reserve mean	Reserve:non-reserve ratio	Source
Leigh MR	1975	7.18	0.45	16	Willis et al. (2003)
Hahei MR	1992	3.15	0.19	16.5	Willis et al. (2003)
Long Bay MR	1995	3.48	0.37	9.4	Ward and Babcock (unpublished data)
Poor Knights MR	1998	12.2	0.76(MK)	16	Denny et al. (2003)
Tawharanui MR	1981	3.5	0.4	8.8	Willis et al. (2003)
Mimiwhangata MP	1982	0.25	0.35	0.71	

Note that MR is no-take marine reserve, MP is marine park, and MK is the Mokohinau Islands, a non-reserve island reference for the Poor Knights.

Species that are targeted by spearfishers were seldom observed in visual transects transects. For example, no blue cod (*Parapercis colias*), three undersize snapper and two porae (*Nemadactylus douglasii*) were observed. 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, personal communication) 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 can lead to overall declines in the mean size and numbers of such species.

No significant difference was found in the overall fish assemblages within and outside the Marine Park using underwater visual census. There were five species significantly more common inside and outside the Marine Park, however, these differences were probably site related, rather than reserve effects, as fishers do not target the majority of these species. Species more common outside the Marine Park were typically schooling fish such as jack mackerel, koheru and sweep. The baited underwater video found that pigfish, leatherjackets and trevally were significantly more common in the Marine Park than in the adjacent control areas. Although Marine Park fishing regulations may protect these species, the Wide Berths in the centre of the Marine Park may simply represent a better habitat for these species than adjacent shallower, and more sheltered coastal waters. The Wide Berths project further out to sea that the rest of the Park and are likely to be influenced by a different current regime and a higher level of wave exposure that the rest of the Park. As expected, plankton feeders, such as demoiselles and trevally were more common in this area. This finding was consistent with the fact that these species are more common at offshore islands like the Poor Knights and Mokohinau, or on the mainland sites with 'offshore' physical characteristics (e.g. Cape Brett) (Kingsford, 1989). Unsurprisingly, both methods found that the deeper reefs in Areas 4 and 5 had significantly more leatherjackets, as deep reefs are their preferred habitat (Ayling, 1981).

As expected, the reef fish assemblage at the Mimiwhangata Marine Park most closely resembled that of the other 'mainland' site Cape Brett. The lower number of species recorded at Mimiwhangata, compared to the other three surveyed areas, was mainly accounted for by low numbers of subtropical wrasse species, common on offshore islands (Denny et al., 2003). This may be because the East Auckland Current, although not having such a heavy influence, does occasionally impinge on the Mimiwhangata coast bringing with it low numbers of subtropical species.

Studies of snapper populations in other coastal marine reserves in northeastern New Zealand have shown a sharp gradient in snapper abundance between no-take areas and adjacent fished areas (Willis et al., 2000, 2003). Gradients of snapper abundance in other coastal marine reserves in northeastern New Zealand suggest fishing effects that extend inside the reserve, rather than spillover effects. At the Cape Rodney to Okakari Point Marine Reserve the peak abundance is in the centre of the reserve, well inside the reserve boundaries (Willis et al., 2000, 2003). It is thus highly unlikely that the lack of contrast between the Mimiwhangata Marine Park and adjacent fished areas is due to the possibility than any effect of protection is being obscured by spillover.

This study demonstrates that the partial closures at Mimiwhangata are ineffective as conservation tools either for heavily targeted species, 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.

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References

Anderson, M.J., 2002. Canonical Analysis of Principal Coordinates (CAP): A Program for Generalised Discriminant Analysis or Canonical Correlation Analysis on the Basis of any Distance Measure. Department of Statistics, University of Auckland. http://www.stat.auckland.ac.nz/~mja/Programs.htm.

Anon, X, 2002. Representative area program. Marine Protected Areas News 3, 4.
Ayling, A.M., 1981. The role of biological disturbance in temperate subtidal encrusting communities. Ecology 830–847.
Ballantine, W.J., Grace, R.V., Doak, W.T., 1973. Mimiwhangata Marine Report. Turbott and Halstead/New Zealand Breweries Limited, Auckland.
Bohnsack, J.A., 1997. Consensus development and the use of marine reserves in the Florida Keys, USA. In: Proceedings of the 8th International Coral Reef Symposium, Panama, pp. 1927–1930.
Brook, F.J., 2002. Biogeography of near-shore fishes in northern New Zealand. Journal of the Royal Society of New Zealand 32, 243–274.
Choat, J.H., Ayling, A.M., 1987. The relationship between habitat structure and fish faunas on New Zealand reefs. Journal of Experimental Marine Biology and Ecology 110, 257–284.
Denny, C.M., Willis, T.J., Babcock, R.C., 2003. Effects of the Poor Knights Islands Marine Reserve on demersal fish populations: II. Report to the Department of Conservation, Science and Research (Grant No. 3270).
Francis, M.P., 2001. Coastal Fishes of New Zealand. An Identification Guide. Reed Publishing Ltd, Auckland.
Francour, P., 1994. Pluriannual analysis of the reserve effect on ichthyofauna in the Scandola natural reserve (Corsica, Northwestern Mediterranean). Oceanologica 17, 309–317.
Francour, P., Harmelin, J.-G., Pollard, D., Sartoretto, S., 2001. A review of marine protected areas in the northwestern Mediterranean region: siting, usage, zonation and management. Aquatic Conservation: Marine and Freshwater Ecosystems 11, 155–188.
Kingsford, M.J., 1989. Distribution patterns of planktivorous reef fish along the coast of northeastern New Zealand. Marine Ecology Progress Series 54, 13–24.
Kingsford, M.J., Battershill, C.N., 1998. Studying Temperate Marine Environments. A Handbook for Ecologists. Canterbury University Press, Christchurch.
Saul, P., Holdsworth, J., 1991. Cooperative Gamefish Tagging in New Zealand Waters, 1975–90. New Zealand Fisheries Technical Report, 25.
Thrush, S.F., Hewitt, J.E., Cummings, V.J., Dayton, P.K., Cryer, M., Turner, S.J., Funnell, G.A., Budd, R.G., Milburn, C.J., Wilkinson, M.R., 1998. Disturbance of the marine benthic habitat by commercial fishing: impacts at the scale of the fishery. Ecological Applications 8, 866–879.
Vacchi, M., Bussotti, S., Guidetti, P., La Mesa, G., 1998. Study of the coastal fish assemblage in the marine reserve of the Ustica Island (southern Tyrrhenian Sea). Italian Journal of Zoology 65, 281–286.
Willis, T.J., Babcock, R.C., 2000. A baited underwater video system for the determination of relative density of carnivorous reef fish. Marine and Freshwater Research 51, 755–763.
Willis, T.J., Denny, C.M., 2000. Effects of the Poor Knights Island Marine Reserve on Demersal Fish Populations. Report to the Department of Conservation, Science and Research (Grant No. 2519).
Willis, T.J., Millar, R.B., Babcock, R.C., 2000. Detection of spatial variability in relative density of fishes: comparison of visual census, angling, and baited underwater video. Marine Ecology Progress Series 198, 249–260.
Willis, T.J., Millar, R.B., Babcock, R.C.Protection of exploited fishes in temperate regions: high density and biomass of snapper Pagrus auratus (Sparidae) in northern New Zealand marine reserves. Journal of Ecological Applications 40, 14–227.
Willis, T.J., Parsons, D.M., Babcock, R.C., 2001. Evidence for long-term site fidelity of snapper (*Pagrus auratus*) within a marine reserve. New Zealand Journal of Marine and Freshwater Research 35, 581–590.