

Effects of marine reserve protection at Goat Island, northern New Zealand

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Abstract The possible effects that marine reserve protection has had on densities of some reef fish and large invertebrates were investigated near Leigh (north-eastern New Zealand) by a series of sampling programmes between 1976 and 1988. Fish counts at intervals during the 6 years after the initial establishment of the Cape Rodney to Okakari Point Marine Reserve in 1975 suggested that red moki (*Cheilodactylus spectabilis*) had increased in abundance whereas five others had remained at approximately constant densities. A comparison of data between 1978 and 1988 also revealed few consistent differences in fish abundances. A detailed survey in 1988 between sites inside and outside the marine reserve showed no clear patterns for sea urchins (*Evechinus chloroticus*) and several fish; trends for increased abundances in the marine reserve of fish such as snapper *Pagrus* (= *Chrysophrys*) *auratus*, blue cod (*Paraperis colias*), and red moki; a very striking increase in numbers of rock lobsters (*Jasus edwardsii*) within the marine reserve; and an obvious trend for increased size of snapper in the marine reserve. Most of the trends, however, were not statistically significant, owing largely to the low power of the tests used. Although it is now generally accepted that the creation of marine reserves such as the one at Leigh results in increased abundances of

certain organisms, our study highlights the difficulty of rigorously demonstrating this, especially for patchily distributed and mobile fish species.

Keywords marine reserve; fish; abundance; rock lobster; sea urchin

INTRODUCTION

Populations of marine organisms may be affected by recruitment variability (Doherty 1982), availability of space (Sale 1979) and/or food (Jones 1986), and fishing (Koslow et al. 1988). Of these, fishing is clearly the factor most directly under human control. Marine reserves offer an opportunity to assess the impact of fishing on wild populations, by protecting areas from exploitation. There are few studies, however, which document the effects of such protection (Bell 1983; Conan 1986; MacDiarmid 1987; McCormick & Choat 1987; Kingsford 1989; Russ & Alcalá 1989).

The Cape Rodney to Okakari Point Marine Reserve near Leigh (Fig. 1) was established in 1975, the first marine reserve in New Zealand (Ballantine 1987). It is regularly visited by large numbers of divers. If the reserve status is a cause for this popularity, it may be desirable to create more marine reserves. A study of recreational use of the marine reserve (New Zealand Department of Lands and Survey 1984) suggested that reserve status might indeed be a contributing factor to human use of the area. There is a widely held view that fish are more abundant in the marine reserve (Ballantine 1987, 1989). In response to a questionnaire (New Zealand Department of Lands and Survey 1984), the majority of respondents gave the opportunity of seeing large numbers of fish and rock lobster as the prime reason for visiting the site.

Although statutory protection from all forms of fishing within the marine reserve has been in place since 1975, there had been a voluntary ban on spear-fishing in the area for 5 years before this. If protection from fishing does have an effect on fish populations, it should be most apparent in this marine reserve.

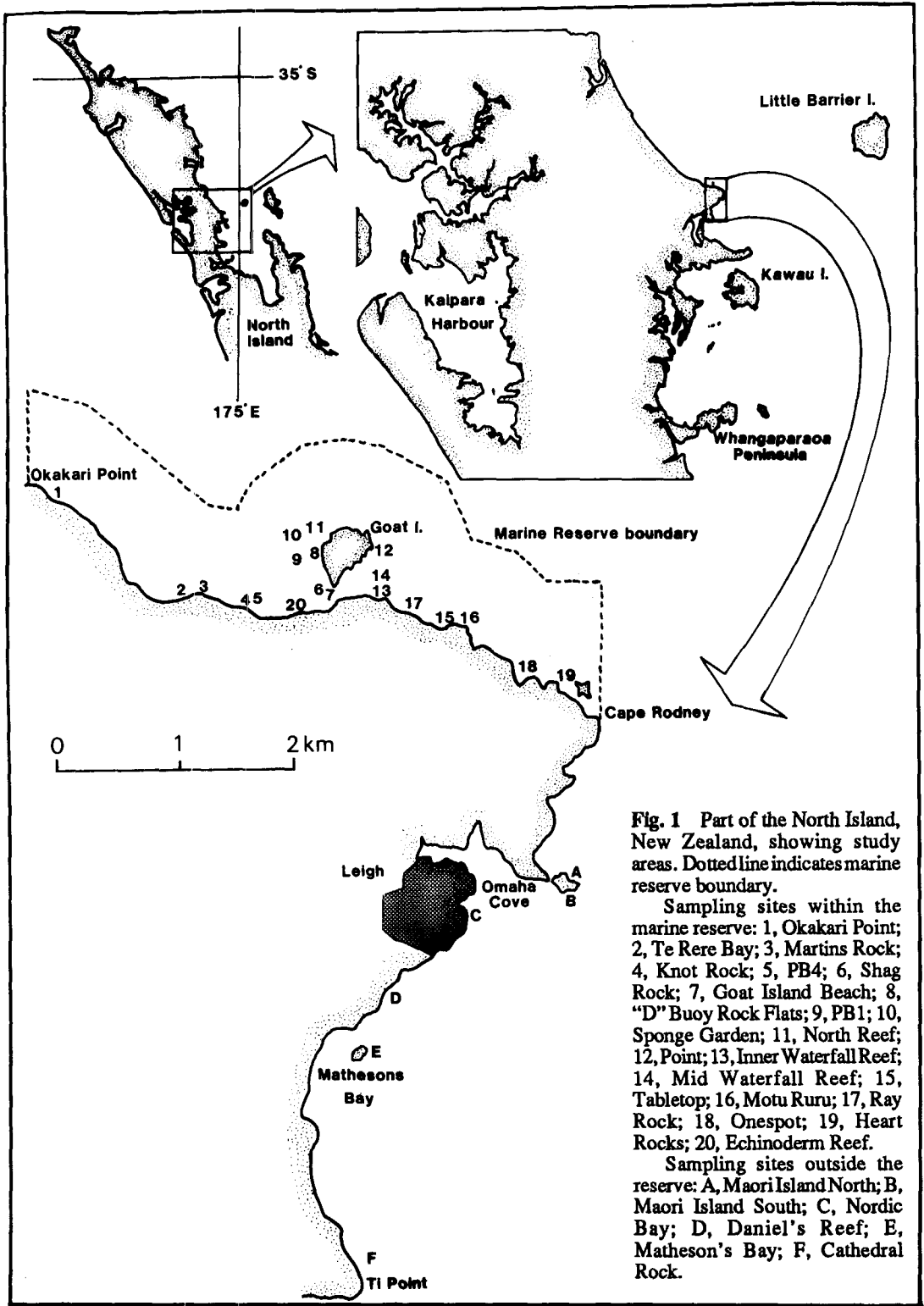


Fig. 1 Part of the North Island, New Zealand, showing study areas. Dotted line indicates marine reserve boundary.

Sampling sites within the marine reserve: 1, Okakari Point; 2, Te Rere Bay; 3, Martins Rock; 4, Knot Rock; 5, PB4; 6, Shag Rock; 7, Goat Island Beach; 8, "D" Buoy Rock Flats; 9, PB1; 10, Sponge Garden; 11, North Reef; 12, Point; 13, Inner Waterfall Reef; 14, Mid Waterfall Reef; 15, Tabletop; 16, Motu Ruru; 17, Ray Rock; 18, Onespot; 19, Heart Rocks; 20, Echinoderm Reef.

Sampling sites outside the reserve: A, Maori Island North; B, Maori Island South; C, Nordic Bay; D, Daniel's Reef; E, Matheson's Bay; F, Cathedral Rock.

Fishing may affect the species composition in an area (e.g., Koslow et al. 1988), the abundance of sought-after species, or the size structure of their populations (e.g., McCormick & Choat 1987). Our study investigated changes in abundances of reef-associated species after the creation of the marine reserve. Differences in abundances of these species between the marine reserve and an adjacent unprotected area were also examined, using a similar, though simplified, design to that used by Kingsford (1989) for planktivorous fish. We predicted that species taken commercially and/or sought by recreational fishermen, e.g., rock lobster (*Jasus edwardsii*), snapper (*Pagrus* (= *Chrysophrys*) *auratus*), red moki (*Cheilodactylus spectabilis*), and blue cod (*Parapercis colias*) would increase in abundance after protection, and that, after 12 years of official protection, these species would be more abundant within the marine reserve than outside.

METHODS

Study area

All sampling was carried out near Leigh, in north-eastern New Zealand (Fig. 1), and comprised three related programmes. First, abundances of several fish species were assessed within the marine reserve intermittently between 1975 and 1982 using randomly selected sites (Table 1). Second, a comparison of fish abundances was made between 1978 and 1988 at 10 geographically fixed sites within the marine reserve (Table 2). Finally,

abundances of several selected species were compared in 1988 (Table 3), between the marine reserve and the adjacent stretch of coast from Cape Rodney to Ti Point (Fig. 1).

The habitats and biota of this area have been described by Ayling (1981) and Choat & Schiel (1982). Briefly, hard substrata down to a depth of c. 4 m are covered by fucal and laminarian algae ("shallow broken rock" habitat). Echinoid-dominated "rock flats" occur between 4 and 10 m. At greater depths, continuous swathes of the laminarian alga *Ecklonia radiata* ("kelp forests") exist.

General sampling methods

All data were recorded on underwater slates while SCUBA diving. All techniques which required estimation of distances were practised before sampling was started. Densities of sea urchins and rock lobsters were estimated using quadrats. Strip transects were used to estimate abundances of a suite of fish which excluded blennioids, pelagic, and planktivorous species. Transects were laid by placing a fibreglass tape on the substratum, a process which often attracted the more inquisitive fish from nearby. After the tape had been laid, therefore, the diver swam off a short distance and waited for the fish attracted to the disturbance to disperse. Details of the methods used for each particular programme are given separately below.

Temporal changes in fish abundance, 1976–82

From December 1975 to February 1982, one of us (A. Ayling) carried out fish counts at seven sites

Table 1 Summary of sampling programme I: Temporal changes in fish abundance, 1976–82.

A. Across sites

Year	Sites
1976	Goat Island Beach, D Buoy, Inner Waterfall, Mid Waterfall
1977	Goat Island Beach, D Buoy, PB1, Inner Waterfall
1978	Goat Island Beach, D Buoy, PB1, Inner Waterfall, Mid Waterfall
1979	Goat Island Beach, D Buoy, PB1, Inner Waterfall, Mid Waterfall
1982	Goat Island Beach, D Buoy, PB1, Inner Waterfall, Mid Waterfall

B. Within sites

Site	Years
Goat Island Beach	1976, 1977, 1978, 1979, 1982
D Buoy	1976, 1977, 1978, 1979, 1982
Sponge Garden	1977, 1978, 1979, 1982
PB1	1976, 1978, 1979, 1982
Point	1976, 1978, 1979, 1982
Inner Waterfall	1976, 1977, 1978, 1979, 1982
Mid Waterfall	1976, 1978, 1979, 1982

within the marine reserve. A varying number of sites were sampled during summer (December–February) each year (Table 1). Abundances were obtained from 50 × 10 m transects by counting fish 5 m either side of a 50 m tape. Sampling was restricted to one habitat at each site and generally five replicate counts were made at each site.

Long-term temporal changes in fish abundance

Fish densities at fixed sites were compared between 1978 and 1988. Fish were counted by A. Ayling in 1978, using 50 × 10 m transects as above, at a number of sites. In 1988, five of the same sites from each of the shallow broken rock and rock flats habitats (see Table 2) were randomly selected and recounted by R. Cole.

Spatial differences in 1988

The sampling for the 1988 comparison was stratified by habitat. The reef was divided into three strata: shallow broken rock, rock flats, and kelp forest. The sites used are given in Table 3. The sampling design for the sea urchins, rock lobsters, and fish species counted within each habitat comprised the fixed factor "Protection", with levels

"Inside" and "Outside" the marine reserve, and, nested within these levels, the random factor "Site". For each habitat three sites were sampled outside the marine reserve, and five within. More sites were sampled within the marine reserve as they formed part of an overlapping sampling programme designed to monitor abundances of organisms in the marine reserve over the next 6 years.

The numbers of sea urchins in 50 haphazardly placed 1 m² quadrats within the shallow broken rock habitat were counted at each site (25 by each diver). The quadrat size of 1 m² was selected to be compatible with previous studies which have assessed sea urchin densities (e.g., Andrew & Choat 1982, 1985; Choat & Schiel 1982).

Numbers of rock lobsters in the shallow broken rock habitat were estimated using 10 × 10 m quadrats as recommended by MacDiarmid (1987) ($n = 12$). Only areas of shallow broken rock with large numbers of boulders and crevices were sampled. A tape was run out 10 m, and a distance of 5 m either side of the tape was estimated. Two divers each exhaustively searched one half of the quadrat using underwater torches, and the counts for each diver's half-quadrat were added.

Table 2 Summary of sampling programme II : Long-term temporal changes 1978–88.

Habitat	Year	Sites
Shallow broken rock	1978	Te Rere Bay, Knot Rock, Shag Rock, Inner Waterfall, Tabletop
	1988	Te Rere Bay, Knot Rock, Shag Rock, Inner Waterfall, Tabletop
Rock flats	1978	Okakari Point, Martins Rock, PB4, North Reef, Motu Ruru
	1988	Okakari Point, Martins Rock, PB4, North Reef, Motu Ruru

Table 3 Summary of sampling programme III : Spatial differences in 1988.

Habitat	Protected sites	Unprotected sites
A. Fish counts		
Shallow broken rock	Knot Rock, Waterfall, Ray Rock, Tabletop, Onespot	Maori I. North, Nordic Bay, Cathedral Rock
Rock flats	Martins Rock, Knot Rock, Echinoderm Reef, Inner North Reef, Waterfall	Maori I. North, Matheson's Bay, Daniel's Reef
Kelp forest	Martins Rock, PB1, Point, Tabletop, Onespot	Maori I. South, Matheson's Bay, Cathedral Rocks
B. Sea urchin counts		
Shallow broken rock	Knot Rock, Waterfall, Ray Rock, Tabletop, Onespot	Maori I. North, Nordic Bay, Cathedral Rock
C. Rock lobster counts		
Shallow broken rock	Martins Rock, Waterfall, Tabletop, Onespot, Heart Rocks	Maori I. North, Daniel's Reef, Cathedral Rock

All fish counts within this programme were made using 25 × 5 m tape transects rather than the 50 × 10 m transects generally used by earlier workers (Choat & Ayling 1987, and references therein). A number of studies have shown that smaller sampling units are more efficient (e.g., Sale & Sharp 1983; McCormick & Choat 1987).

Counts were made by only one diver (R. Cole). The diver counting the fish swam the free end of the tape out, while the other diver held the reel. When 25 m had been swum out, this diver tugged on the tape to indicate that the appropriate distance had been reached. The area 2.5 m either side of the diver's path was scanned for fish, both on and just above the substratum, while the tape was swum out. Caves, ledges, and overhangs were searched for cryptic individuals in rugose areas. Counts were made between 0800 and 1700 h, and restricted to periods when underwater visibility was greater than 6 m. Individual fish which exhibited diver-oriented behaviour were excluded from counts.

A different sampling technique was required in the kelp forest habitat, where the density of the algae prevented the full 5 m being scanned. Sampling in this habitat was done in two stages; one above and one below the kelp. As the diver swam the tape out, he counted all snapper and spotties (*Notolabrus celidotus*) above the kelp. These species were the two most attracted to divers, and an index of their abundance could only be derived by counting as the tape was laid, as in the other habitats. The end of the tape was then tied to a kelp plant, and the diver turned and pushed his way back through the kelp, an estimated 1.25 m from the tape, counting fish between his path and the tape, and an estimated 1.25 m beyond the path away from the tape. On reaching the beginning of the tape, the diver turned and repeated the procedure for the other side of the tape. When the tape had been run out the required distance, the assisting diver would move back at least 5 m from the end of the transect to avoid attracting fish. Snapper and spotties were not counted during the substratum-searching part of the transect.

Snapper were chosen for a comparison of size-frequencies between protected and unprotected sites because they form the basis of the most important commercial and recreational fishery in the Hauraki Gulf. The sizes of all snapper (in 10 cm classes) which passed through a 2 × 2 × 2 m cube of water during a 10-min observation period were estimated. Comparisons were made between nine sites in the rock flats habitat inside and outside the marine reserve, spread over 2 days in February 1988.

RESULTS

Temporal changes in fish abundance, 1976–82

The abundances of six species of common reef fish were pooled across shallow broken rock, rock flats, and kelp forest sites (Fig. 2). These six species occurred at a sufficient number of sites for comparisons between years to be made. Numbers of red moki increased between 1976 and 1982. Goatfish (*Upeneichthys lineatus*), blue cod, and spotty abundances showed no clear trend with time. The abundance of snapper increased from 1976 to 1979, but then decreased again in 1982. The abundance of leatherjacket (*Parika scaber*) decreased between 1976 and 1978, and then increased through 1979 to 1982.

Table 4 gives Spearman rank correlation coefficients between mean abundance and year within sites, for each of the six species. Four of the 40 comparisons are statistically significant. Red moki and blue cod at Mid Waterfall rock flats, and blue cod at the Point, showed increasing abundances through time, whereas leatherjacket at Inner Waterfall shallow broken rock showed a significant negative relationship with time. Of the 40 comparisons, nine had negative Spearman's coefficients, one had a coefficient of zero, and the remaining 30 showed positive relationships.

Long-term temporal changes in fish abundance

The comparison of sites between 1978 and 1988 did not show consistent differences for any species (Table 5). In the shallow broken rock habitat, the abundances of goatfish and hiwihiwi (*Chironemus marmoratus*) seemed to have decreased. However, neither these, nor the trend for the density of spotties on rock flats to have increased over the 10-year period, were consistent across all sites.

Spatial differences in 1988

DENSITIES OF INVERTEBRATES

Densities of sea urchins could not be statistically analysed as a result of intransigently heterogeneous variances. Densities were generally higher within the marine reserve, however, and numbers were very much lower at one site outside the marine reserve (Fig. 3).

Many more rock lobsters were found inside the marine reserve than outside; rigorous searching of three sites outside the marine reserve containing suitable habitat failed to reveal any individuals of this species. The mean density of rock lobsters

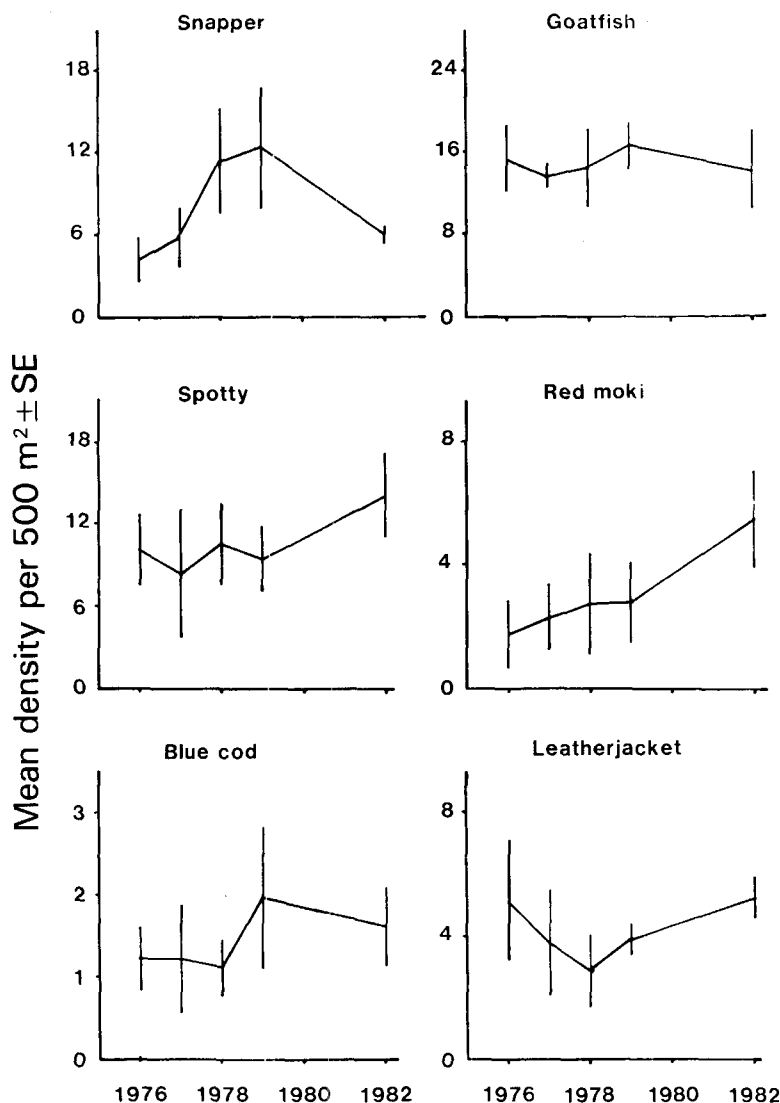


Fig. 2 Mean densities of six species of fish pooled across varying numbers of sites for the years 1976–82.

within the marine reserve was 5.25 (SEM = 0.97, $n = 60$), but abundances were highly variable at the replicate (within site) level (mean ratio SEM/mean \pm standard deviation = 0.44 ± 0.19).

FISH DENSITIES

Table 6 lists the species of fish counted in this programme. More fish species occurred at sites within the marine reserve than outside it (Fig. 4), but the number of all species counted at any of the sites was generally low.

Densities of fish were also low, although goatfish were present in high densities in kelp forests, and spotties were common in the shallow broken rock

and rock flats habitats. Standard errors were generally high relative to means (Table 7), reflecting small-scale variability in patterns of distribution.

Results of ANOVA are not presented as only 6 of 26 comparisons could be transformed to homoscedasticity. Mean densities in each habitat inside and outside the marine reserve are compared in Table 8. In the shallow broken rock habitat, no snapper, leatherjacket, or butterflyfish (*Odax pullus*) were counted outside the marine reserve, and there were higher mean densities of red moki in the marine reserve. Parore (*Girella tricuspidata*) were more abundant outside the marine reserve. On rock flats, no red moki were counted outside the marine reserve.

Snapper, blue cod, and red moki tended to be more abundant in kelp forests at protected sites. Goatfish occurred at higher mean densities in kelp forests outside the marine reserve. Of the 26 protection comparisons in this programme, 13 had non-overlapping standard error bars with greater values of the mean inside the marine reserve, nine had overlapping standard error bars, and four had non-overlapping standard error bars with greater values of the mean outside the marine reserve.

Inter-site variability was very pronounced for many species in most habitats. Densities of snapper,

goatfish, leatherjacket, parore, banded wrasse (*Notolabrus fucicola*), hiwihiwi, and butterfish were absent from one or more sites in the shallow broken rock habitat (Fig. 5), as were leatherjacket, red moki, banded wrasse, and hiwihiwi in the rock flats habitat (Fig. 6), and snapper, leatherjacket, blue cod, red moki, banded wrasse and hiwihiwi in the kelp forest habitat (Fig. 7).

The comparison of snapper size-frequencies shows that a diver can expect to see larger snapper in greater abundances in the marine reserve than outside it (Fig. 8).

Table 4 Spearman rank correlation coefficients between mean abundance of six species of fish and year, for seven sites. -, insufficient data to analyse; *, significant correlation at $P = 0.05$.

Site	Snapper	Goatfish	Red moki	Spotty	Blue cod	Leatherjacket
Goat Island Beach ($n = 4$)	-	0.8	0.825	0.6	0.825	0.3
Mid Waterfall ($n = 4$)	0.6	0.8	0.95*	0.35	0.95*	-0.2
D Buoy Rock Flats ($n = 5$)	-	0.1	0.225	0.4	0.45	0.6
Sponge Garden ($n = 4$)	0.2	0.4	0.75	-0.4	0.4	-0.8
Inner Waterfall ($n = 5$)	-0.6*	-0.5	0.875	0	-0.825	-1.0*
PB1 ($n = 4$)	0.4	-0.8	0.75	0.8	0.8	0.8
Point ($n = 4$)	-0.2	0.2	0.4	0.4	1.0*	0.4

* $n = 4$

Table 5 Mean densities of fish (\pm standard error) in 50×10 m transects at five shallow broken rock and five rock flat sites in the marine reserve. 1988 densities of spotty and snapper have been converted from abundances per 250 m^2 to abundances per 500 m^2 , and are therefore expressed without standard errors. Asterisks indicate likely changes in abundance.

A. Shallow Broken Rock

	Te Rere Bay		Knot Rock		Inner Waterfall		Tabletop		Shag Rock	
	1978	1988	1978	1988	1978	1988	1978	1988	1978	1988
Female spotty	25.8 \pm 4.4	19.2	32.8 \pm 3.1	0.0	3.0 \pm 1.5	26.0	23.6 \pm 1.8	31.2		
Male spotty	3.2 \pm 0.4	2.4	3.4 \pm 0.7	1.6	5.8 \pm 1.1	4.0	2.6 \pm 0.7	2.4		
Snapper	1.8 \pm 0.7	0	8.4 \pm 3.7	0.8	1.6 \pm 0.2	5.2	0.8 \pm 0.4	0.4	21.5 \pm 3.2	9.6
Goatfish	5.0 \pm 1.8	1.4 \pm 0.9	8.2 \pm 3.1	1.2 \pm 0.4	7.2 \pm 2.0	1.0 \pm 0.3*	0.8 \pm 0.6	0.6 \pm 0.4	11.7 \pm 3.2	3.0 \pm 0.3*
Leatherjacket	0.8 \pm 0.4	0.4 \pm 0.4	3.8 \pm 0.9	0.6 \pm 0.2	5.6 \pm 0.6	2.0 \pm 0.5	3.2 \pm 0.6	1.6 \pm 0.7	0.7 \pm 0.3	4.4 \pm 1.3*
Blue cod	1.2 \pm 0.7	1.2 \pm 0.7	0.2 \pm 0.2	2.2 \pm 0.7	1.6 \pm 0.7	2.0 \pm 0.7	0.8 \pm 0.2	0.8 \pm 0.2	0.4 \pm 0.2	0.6 \pm 0.4
Hiwihiwi	14.8 \pm 5.9	1.8 \pm 0.5*	16.0 \pm 1.9	1.0 \pm 0.3*	16.4 \pm 3.0	1.4 \pm 0.7*	20.2 \pm 4.2	3.2 \pm 1.0*	6.5 \pm 2.4	4.2 \pm 1.7
Red moki	6.8 \pm 2.5	2.8 \pm 0.6	11.6 \pm 2.8	1.4 \pm 0.6*	9.0 \pm 1.8	1.6 \pm 0.9	6.6 \pm 2.3	0.8 \pm 0.6	0.8 \pm 0.3	1.8 \pm 0.6
Parore	4.2 \pm 1.8	1.0 \pm 0.6	1.4 \pm 0.5	0.8 \pm 0.4	5.0 \pm 1.5	0.2 \pm 0.2	53.7 \pm 7.7	13.2 \pm 8.2	0.7 \pm 0.3	3.8 \pm 1.5
Silver drummer	0.0 \pm 0.0	0.2 \pm 0.2	0.0 \pm 0.0	0.0 \pm 0.0	0.4 \pm 0.4	0.0 \pm 0.0	4.8 \pm 1.7	1.6 \pm 0.9	0.0 \pm 0.0	0.8 \pm 0.8
Marblefish	0.6 \pm 0.4	0.4 \pm 0.2	0.2 \pm 0.2	0.0 \pm 0.0	1.0 \pm 0.6	0.0 \pm 0.0	3.6 \pm 0.9	0.2 \pm 0.2	0.1 \pm 0.1	0.4 \pm 0.2

B. Rock flats

	Martins Rock		PB4		North Reef		Motu Ruru		Okakari Point	
	1978	1988	1978	1988	1978	1988	1978	1988	1978	1988
Female spotty	6.6 \pm 1.9	22.8	6.0 \pm 1.0	6.4	2.2 \pm 0.6	14.0	11.0 \pm 1.5	13.6	9.0 \pm 1.3	25.6
Male spotty	1.8 \pm 0.7	2.8	0.8 \pm 0.2	1.2	1.0 \pm 0.3	2.0	1.5 \pm 0.6	1.2	2.2 \pm 0.7	2.4
Snapper	17.0 \pm 4.3	3.6	20.0 \pm 5.0	10.8	7.4 \pm 1.6	0.8	7.0 \pm 5.1	0.4*	14.0 \pm 8.2	0.4
Goatfish	13.0 \pm 3.6	5.4 \pm 1.5	9.6 \pm 2.5	1.8 \pm 0.8	7.6 \pm 2.1	4.0 \pm 2.9	10.3 \pm 2.7	0.0 \pm 0.0*	8.0 \pm 4.6	6.8 \pm 1.9
Leatherjacket	0.8 \pm 0.4	1.0 \pm 0.3	3.6 \pm 1.2	0.6 \pm 0.4	3.4 \pm 0.5	4.0 \pm 1.1	2.2 \pm 0.9	4.0 \pm 1.8	0.0 \pm 0.0	0.2 \pm 0.2
Blue cod	3.4 \pm 1.2	3.2 \pm 0.2	4.0 \pm 1.1	7.2 \pm 1.5	1.8 \pm 0.5	2.2 \pm 1.0	1.2 \pm 1.0	1.2 \pm 0.7	6.6 \pm 0.9	4.8 \pm 0.4
Hiwihiwi	1.6 \pm 0.8	0.8 \pm 0.4	0.0 \pm 0.0	0.4 \pm 0.4	2.0 \pm 0.7	2.4 \pm 0.5	2.0 \pm 0.4	1.6 \pm 0.8	0.4 \pm 0.0	0.0 \pm 0.0
Red moki	2.4 \pm 1.1	3.4 \pm 0.9	0.0 \pm 0.0	0.6 \pm 0.4	0.2 \pm 0.2	4.0 \pm 0.7	1.8 \pm 0.9	4.8 \pm 1.9	2.4 \pm 0.8	0.6 \pm 0.4

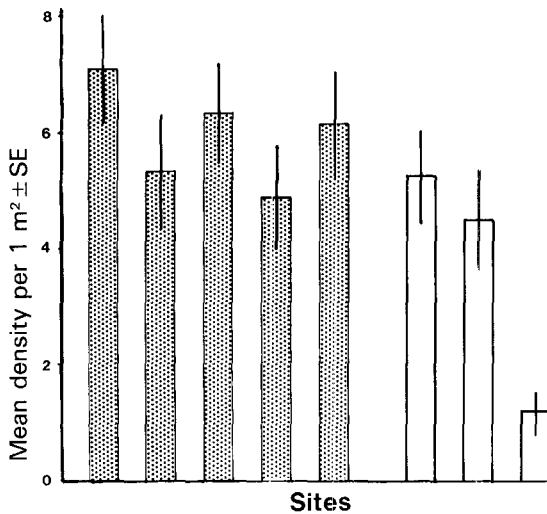


Fig. 3 Mean densities of sea urchins, *Evechinus chloroticus*, at sites inside and outside the marine reserve \pm standard error ($n = 50$). Sites in the marine reserve, shaded bars left to right: Tabletop; Knot Rock; Onespots; Ray Rock; Waterfall. Sites outside the marine reserve, open bars left to right: Cathedral Rock; Maori Island North; Nordic Bay.

Table 6 Species counted in fish transects, 1988.

Spotty	<i>Notolabrus celidotus</i> (Bloch & Schneider, 1801)
Snapper	<i>Pagrus (= Chrysophrys) auratus</i> (Bloch & Schneider, 1801)
Goatfish	<i>Upeneichthys lineatus</i> (Bloch & Schneider, 1801)
Leatherjacket	<i>Parika scaber</i> (Bloch & Schneider, 1801)
Blue cod	<i>Parapercis colias</i> (Bloch & Schneider, 1801)
Banded wrasse	<i>Notolabrus fucicola</i> (Richardson, 1840)
Hiwihiwi	<i>Chironemus marmoratus</i> Günther, 1860
Red moki	<i>Cheilodactylus spectabilis</i> (Hutton, 1872)
Butterfish	<i>Odx pullus</i> (Bloch & Schneider, 1801)
Parore	<i>Girella tricuspidata</i> (Quoy & Gaimard, 1824)
Silver drummer	<i>Kyphosus sydneyanus</i> (Günther, 1886)
Marblefish	<i>Aplodactylus arctidens</i> Richardson, 1839
John dory	<i>Zeus faber</i> Linnaeus, 1758
Scarlet wrasse	<i>Pseudolabrus miles</i> (Bloch & Schneider, 1801)
Sandager's wrasse	<i>Coris sandageri</i> (Hector, 1884)

DISCUSSION

Temporal changes in fish abundance, 1976–82

A greater number of positive relationships between year and mean abundance were found than would be expected by chance. Also, when the data were pooled across sites, there were more positive trends with time than expected. The abundance of red moki clearly increased through time. The failure to detect an increase in abundances of snapper may be a result of the summer influx of small individuals into the rock flats habitat (Kingett & Choat 1981). This would not be expected to reflect fishing activity (Rothschild 1986). Alternatively, it may be that the effect of protection was already established in 1976 when the first comprehensive counts were made. Problems in obtaining accurate counts may also have arisen because of diver-positive behaviour of this species. Snapper were not counted at several sites because of this problem.

Long-term temporal changes in fish abundance

No consistent differences in fish density were detected between 1978 and 1988. Possible reasons for this are: first, there have been no changes during this period; second, differences between observers have obscured any patterns present; third, the protection effect was already established in 1978, 3 years after the marine reserve was officially created; fourth, changes in the distribution of habitats have occurred during the 10-year interval, which has led to inconsistent changes in numbers; and finally, small-scale variability (e.g., between days or months) was of sufficient magnitude to obscure any real patterns. The last four possible explanations are neither mutually exclusive nor directly testable. Patterns of movement which would affect small-scale variability, however, may be important, and ongoing studies are investigating this aspect of fish ecology in the marine reserve (G. P. Jones pers. comm.).

Spatial comparison

The abundance of sea urchins outside the marine reserve was greatly influenced by the low densities at one unprotected site. This may reflect human activity, chance recruitment events, or some unknown, unfavourable site characteristic. There was no evidence of an effect of fish predation on densities of urchins within the marine reserve (see Andrew & Choat 1982).

There was a clear difference in the densities of rock lobsters between sites inside and outside the

Fig. 4 Mean number of fish species counted inside (shades) and outside (clear) the reserve \pm standard error, for three habitats. Shallow broken rock In, $n = 30$; Out, $n = 17$. Rock flats In, $n = 30$; Out, $n = 18$. Kelp forest In, $n = 25$; Out, $n = 15$.

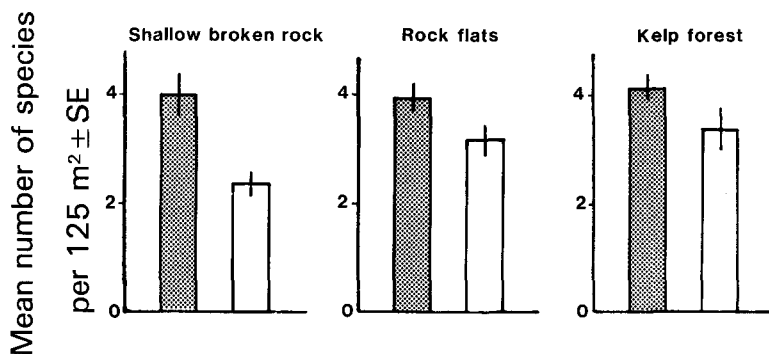
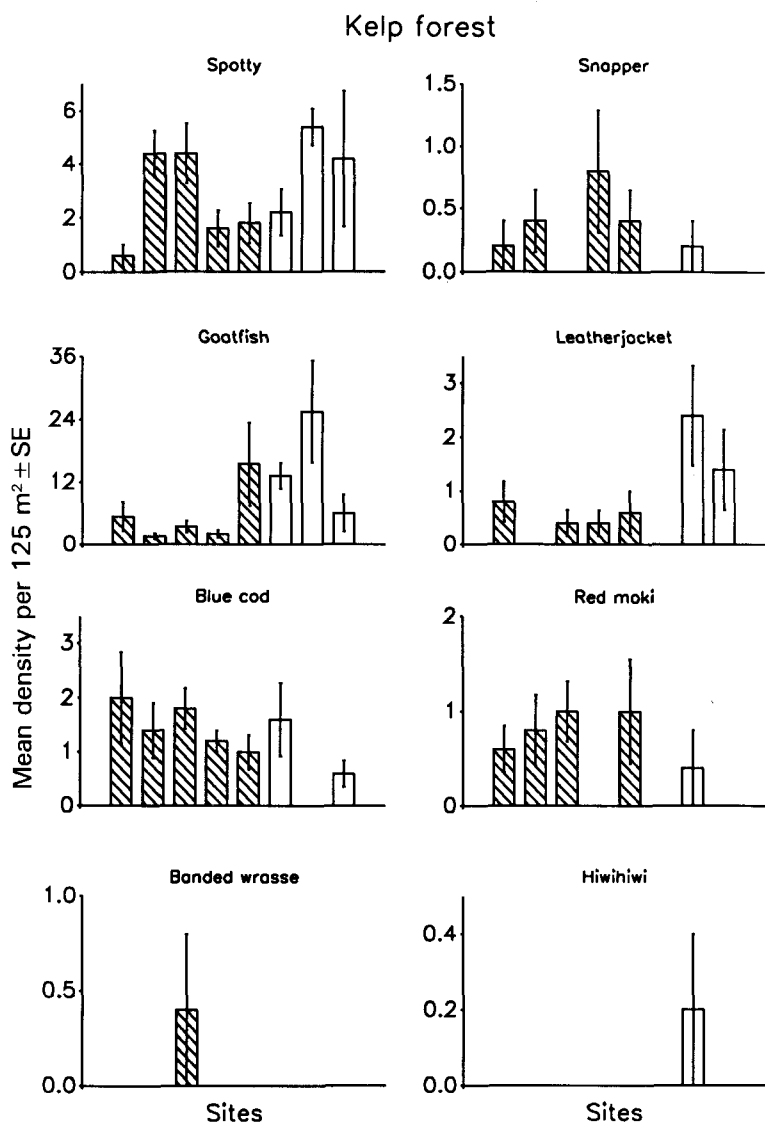


Fig. 5 Mean densities of fish counted in the kelp forest habitat inside and outside the marine reserve \pm standard error. Sites in the marine reserve, shaded bars left to right: PB1, Martins Rock, Onespot, Point, Tabletop; sites outside the marine reserve, open bars left to right: Cathedral Rock, Maori Island South, Matheson's Bay.



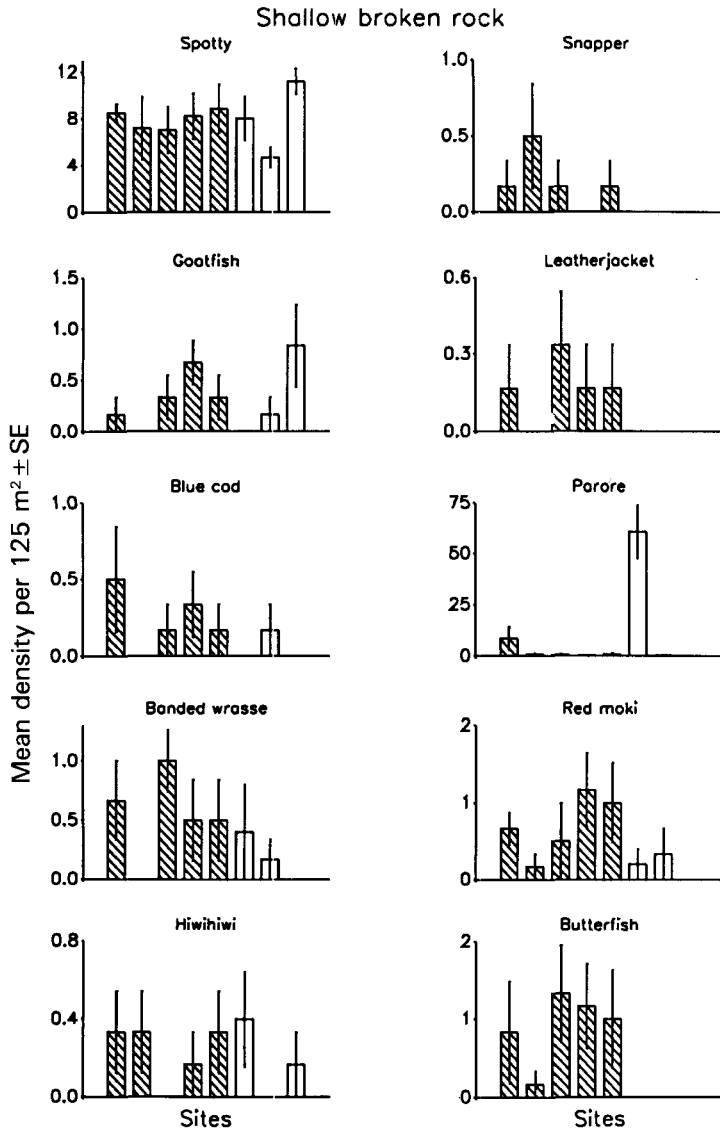


Fig. 6 Mean densities of fish counted in the shallow broken rock habitat at sites inside and outside the marine reserve ± standard error. Sites in the marine reserve, shaded bars left to right: Tabletop, Knot Rock, Onespot, Ray Rock, Inner Waterfall; sites outside the marine reserve, open bars left to right: Cathedral Rock, Maori Island North, Nordic Bay.

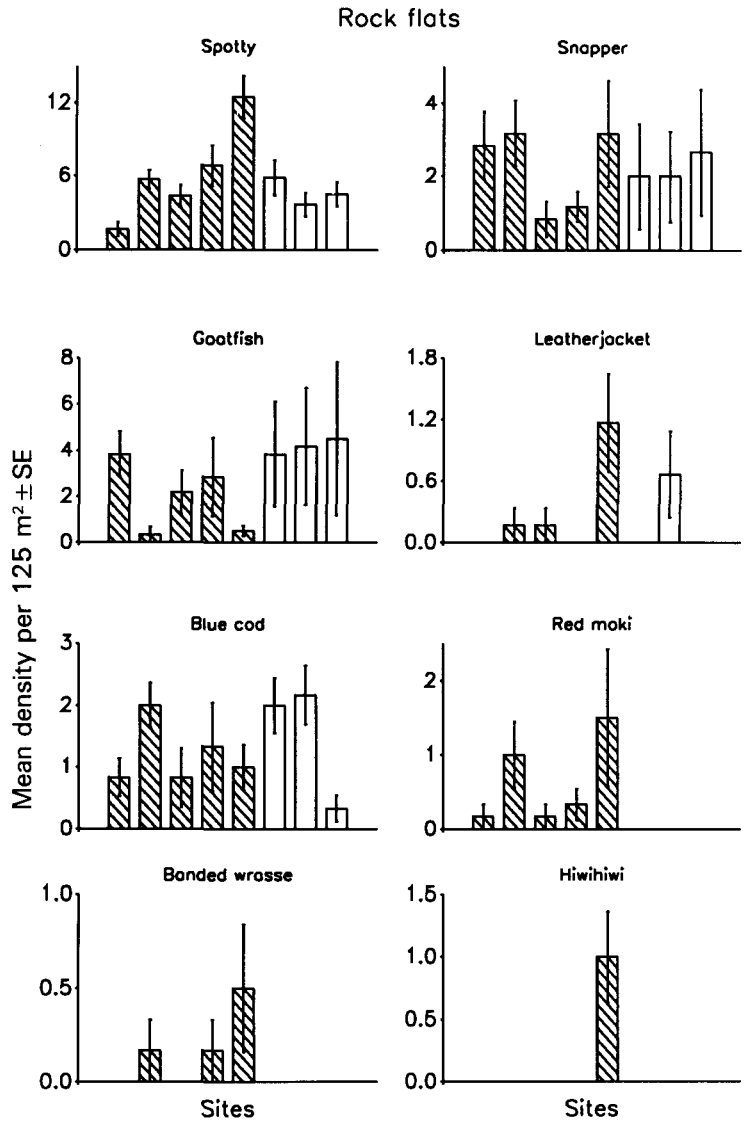
marine reserve. Despite rigorous searching, no rock lobsters were found when sampling outside the marine reserve. This may reflect a behavioural response to hunting pressure; the individuals may have been present, but more cryptic than in the protected sites. This hypothesis is supported by a smaller average size of rock lobsters outside the marine reserve (A. B. MacDiarmid pers. comm.).

We interpret the patterns of goatfish abundance as resulting from site characteristics other than exploitation. This species is not regularly captured

by line or net, and it reaches highest densities in sandy gutters in deeper water (Ayling 1978), which we did not sample.

The failure to detect differences in snapper density in the rock flats habitat is probably the result of the seasonal influx of 0⁺, 1⁺, and 2⁺ fish into this habitat (Kingett & Choat 1981); sampling in a different season could reveal differences in densities. High densities of juveniles may have also obscured real differences in populations of blue cod in this habitat. Information on the size structure of

Fig. 7 Mean densities of fish counted in the rock flats habitat inside and outside the marine reserve \pm standard error. Sites in the marine reserve, shaded bars left to right: Echinoderm Reef, Inner North Reef, Knot Rock, Martins Rock, Mid Waterfall; sites outside the marine reserve, open bars left to right: Matheson's Bay, Daniel's Reef, Maori Island North.



populations of blue cod inside and outside the marine reserve would allow this recruitment effect to be assessed.

The size-frequency data for snapper suggested differences between protected and unprotected areas beyond those in the abundance data. Snapper in the marine reserve were generally larger than in the unprotected area. Again we interpret this as reflecting fishing pressure outside the marine reserve. Small snapper were more abundant outside the marine reserve. Whether this is a result of chance patterns of

habitat occupation, of choice by small snapper for more favourable feeding areas, or of mortality from piscivores within the marine reserve is unknown.

Size-frequency estimation during the transect counts was not attempted, as it was found that the diver's full attention was required just to count fish of all the species being sampled. However, future studies of the effects of protection should sample only a few sought-after species, and estimate sizes of these species during counts. In this manner, effects of fishing methods which are size-selective, such as

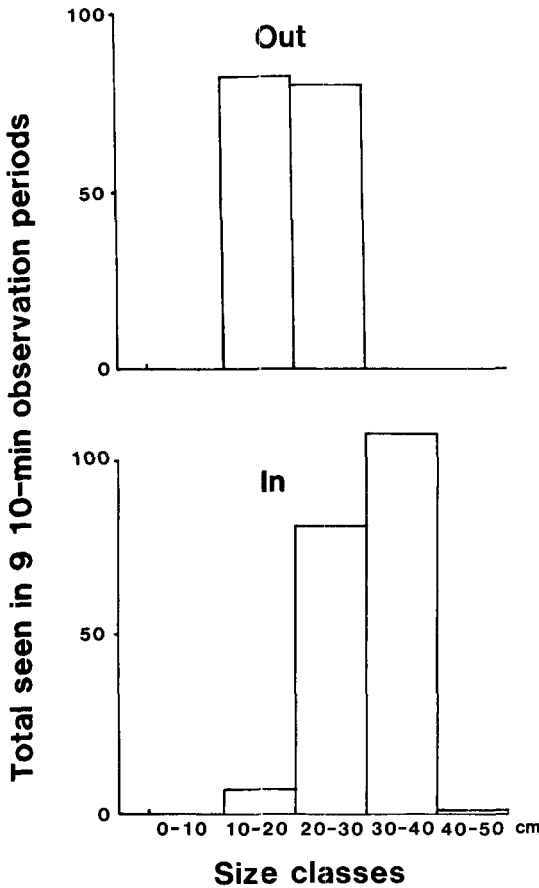


Fig. 8 Size-frequency distributions of snapper, *Pagrus* (= *Chrysophrys*) *auratus* on the rock flats habitat, inside and outside the marine reserve.

spearfishing, may be detected. Also, adult populations may be compared without the seasonal influence of recruitment, which is unlikely to reflect fishing activity (e.g., Rothschild 1986).

Although the inside/outside comparison suggests a protection effect for rock lobster, butterfish, and perhaps red moki, the sampling design is pseudo-replicated (*sensu* Hurlbert 1984). The lack of other protected sites prevents a true assessment of the effects of protection. However, these data, the temporal comparison, and other studies (e.g.,

Table 7 Precision (standard error \times 100/mean) for fish counts in three habitats (SBR, shallow broken rock; RF, rock flats; KF, kelp forest), in protected and unprotected areas.

	Habitat					
	SBR		RF		KF	
Protection	In	Out	In	Out	In	Out
<i>n</i>	30	17	30	18	25	15
Species						
Spotty	11	12	13	14	18	23
Snapper	44	—	19	36	35	100
Goatfish	28	48	25	36	34	26
Leatherjacket	41	—	43	69	30	36
Blue cod	39	100	18	20	15	39
Parore	55	41	47	100	100	100
Banded wrasse	25	73	51	—	100	—
Red moki	26	73	35	—	24	100
Hiwihiwi	34	54	50	—	—	100
Butterfish	—	—	—	—	—	—
Silver drummer	74	—	—	—	—	—
Marblefish	100	100	—	—	—	—
Scarlet wrasse	—	—	—	—	50	100

Table 8 Means and standard errors for fish counts in three habitats (SBR, shallow broken rock; RF, rock flats; KF, kelp forest), in protected and unprotected areas, and probabilities of H_0 being true as a result of chance from *t*-tests. —, no individuals of this species counted in this habitat; *, *t*-test for unequal variances.

Species	Habitat								
	SBR			RF			KF		
	In <i>n</i> = 30 Mean \pm SE	Out <i>n</i> = 17 Mean \pm SE	Prob.	In <i>n</i> = 30 Mean \pm SE	Out <i>n</i> = 18 Mean \pm SE	Prob.	In <i>n</i> = 25 Mean \pm SE	Out <i>n</i> = 15 Mean \pm SE	Prob.
Spotty	7.93 \pm 0.84	7.94 \pm 0.96	0.995	6.20 \pm 0.84	4.67 \pm 0.66	0.156	2.56 \pm 0.45	3.93 \pm 0.92	0.196
Snapper	0.20 \pm 0.09	0.00 \pm 0.00	0.031*	2.23 \pm 0.43	2.22 \pm 0.81	0.989	0.36 \pm 0.13	0.07 \pm 0.07	0.049*
Goatfish	0.30 \pm 0.09	0.35 \pm 0.17	0.758	1.93 \pm 0.48	4.17 \pm 1.49	0.169*	5.56 \pm 1.87	14.87 \pm 3.91	0.044*
Leatherjacket	0.17 \pm 0.07	0.00 \pm 0.00	0.023	0.30 \pm 0.13	0.22 \pm 0.15	0.704	0.44 \pm 0.13	1.27 \pm 0.45	0.098*
Blue cod	0.23 \pm 0.09	0.06 \pm 0.06	0.117*	1.20 \pm 0.21	1.50 \pm 0.29	0.403	1.48 \pm 0.22	0.73 \pm 0.28	0.043
Parore	2.23 \pm 1.23	18.82 \pm 7.64	0.047*	—	—	—	—	—	—
Banded wrasse	0.53 \pm 0.13	0.18 \pm 0.13	0.084	0.17 \pm 0.08	0.00 \pm 0.00	0.057	0.08 \pm 0.08	0.00 \pm 0.00	0.327
Red moki	0.70 \pm 0.18	0.18 \pm 0.13	0.022*	0.63 \pm 0.22	0.00 \pm 0.00	0.008	0.68 \pm 0.16	0.13 \pm 0.13	0.023
Hiwihiwi	0.23 \pm 0.08	0.18 \pm 0.10	0.656	0.20 \pm 0.10	0.00 \pm 0.00	0.056	0.00 \pm 0.00	0.07 \pm 0.07	0.334
Butterfish	0.90 \pm 0.24	0.00 \pm 0.00	0.001	—	—	—	—	—	—

MacDiarmid 1987; McCormick & Choat 1987) combine to provide a credible case for an effect of protection.

Studies of the behaviour of people involved in gathering organisms for recreation, as well as commercially, would contribute to our understanding of the effects of protection. Data on the species, number and size of animals removed, and patterns of harvesting in time and space would allow a clearer interpretation of the patterns of abundance we have outlined. Long-term data on recruitment in protected and unprotected areas would also greatly further our understanding of the factors determining the abundance of marine organisms.

The data above suggest that marine reserves may affect the local abundances of certain species of marine animals, but not others. The effects are most strongly demonstrated for highly sought or vulnerable species, particularly sedentary ones. The mobility and patchy distribution of other species probably contributed to the lack of a statistically significant protection effect, and many more sites would be required to adequately evaluate differences in the abundances of these species.

ACKNOWLEDGMENTS

M. P. Francis, D. R. Schiel, C. Syms and two anonymous referees gave helpful comments on the manuscript. Without W. J. Ballantine there would have been no marine reserve to study. Funding for part of this study was provided by the Department of Conservation. J. Ackley, F. Crouch, K. Denyer, B. Hartill, M. Morrison, A. Pryce, S. Scott, K. Tricklebank, S. White, and especially S. Connell helped with diving. G. P. Jones commented on survey design, and L. Planck gave statistical advice.

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