WK-14233-RES

TE WHANGANUI A HEI MARINE RESERVE FISH MONITORING 2004: FINAL REPORT

AUCKLAND UNISERVICES LIMITED

A wholly owned company of

THE UNIVERSITY OF AUCKLAND

Prepared for:

Mr Jason Roxburgh
Hauraki Area Office
Department of Conservation
PO Box 343
Thames

Prepared by:

Dr Richard B. Taylor[1]
Dr Marti J. Anderson[2]
N. Usmar[1]
Dr Trevor J. Willis
Leigh Marine Laboratory[1]
Department of Statistics[2]
University of Auckland
Tel: 64-9-422-6111
Fax: 64-9-422-6113
Email: rb.taylor@auckland.ac.nz

Date: July 2004

TABLE OF CONTENTS

EXECUTIVE SUMMARY

- This report describes the results of a survey of fish abundances in the Te Whanganui a Hei (or Hahei) Marine Reserve, northeastern New Zealand. The survey was undertaken in June 2004 and continues a time-series that started in 1997.

- In line with previous estimates made using baited underwater video (BUV), legal-sized snapper (*Pagrus auratus*) were 18.3 times more abundant inside the reserve than on the adjacent unprotected coast. An increase in this ratio from 9.0:1 in 2003 was due to a decline in fish numbers outside the reserve rather than an increase inside. Snapper inside the reserve continue to be significantly larger than those outside (by an average of 89 mm fork length).

- Densities of legal-sized snapper recorded in the Hahei reserve using BUV appear to have stabilised at less than half the densities recorded in two other northeastern New Zealand marine reserves (the Cape Rodney to Okakari Point Marine Reserve, and the Poor Knights Islands Marine Reserve).

- Blue cod (*Parapercis colias*) were present at relatively low densities inside and outside the reserve, with the trend for higher densities to occur within the reserve continuing in 2004 (based on BUV data).

- Underwater visual censuses (UVC) continue to indicate that marine reserve protection has no or weak effects on the majority of other fish species at Hahei. Species richness and total numbers of individuals appear to be responding more strongly to the environmental gradient running through the reserve than to reserve status.

Recommendations

- The fish monitoring programme should be continued with the current levels of sample replication regarded as a minimum level of effort, but the Department of Conservation may wish to consider reducing the frequency of surveys (perhaps to every second year) given that densities of snapper and blue cod have been relatively stable over the past few years, and that no rapid changes are occurring in the rest of the fish assemblage. If this is done surveys should still be carried out at the same time of year to ensure comparability among years.

- The Department should consider initiating the mapping of benthic habitats and the monitoring of important benthic organisms inside and outside the reserve. This would allow for detection of changes in ecosystem-level changes due to, for example, increased sedimentation, or trophic cascades resulting from the effects of higher numbers of predatory fishes and lobsters on sea urchins and in turn seaweeds. Habitat change may also help to explain temporal changes in abundances of many of the fish species currently monitored.

INTRODUCTION

The monitoring of marine reserves has three related, but distinctive functions. First, long-term datasets can be used to examine responses of populations under reserve protection relative to fished areas. Second, they allow an assessment of the natural variability associated with species abundance in particular locations, which provides a context for subsequent changes. These changes might occur as a result of sudden (pulse) disturbances or gradual (press) changes, either of which may be of natural or anthropogenic origin. Third, long-term monitoring data assist in the interpretation of environmental changes arising indirectly from changes in the relative density of predators (trophic cascades).

The Te Whanganui a Hei (or Hahei) Marine Reserve was gazetted in 1993. Regular monitoring of the abundance of reef fishes commenced in 1997 (Willis et al. 2003a) and has continued on an annual or semi-annual basis until the present time. As a relatively young reserve, monitoring at Hahei has enabled the assessment of change in fish density soon after reserve regulations took effect, whereas other reserves monitored by the current programme are considerably older (Cape Rodney to Okakari Point Marine Reserve, gazetted 1975; Tawharanui Marine Park, established 1982). In the absence of comparable data collected prior to reserve establishment, comparison of trends in fish numbers within reserves of varying age is our best opportunity to determine rates of recovery of depleted fish species in protected areas.

Assessment of the distribution and relative abundance of species and assemblage structure in the Hahei marine reserve and adjacent fished areas is complicated by environmental gradients found in this area. The western reserve and reference areas are characterised by relatively calm, shallow and turbid waters with a coastal/estuarine influence. To the east, wave-exposed, deep, clear waters are more characteristic (Shears et al. 2000).

Fish monitoring at Hahei has been done using two separate, but concurrently run, methodologies. Carnivorous fishes that are commonly exploited by fishers were surveyed using baited underwater video (BUV: Willis & Babcock 2000, Willis et al. 2000). This method allows the collection of both relative density and size data from species (especially the snapper *Pagrus auratus*) not amenable to sampling using traditional diver census methods. The remainder of the demersal reef species were surveyed using underwater visual census (UVC) transects. The complete survey programme to date is listed in Table 1.

This report presents the results of a single survey conducted in June 2004 using identical techniques to previous years (except for the addition of a second camera system). It should be read in conjunction with the previous report and others cited therein (Taylor et al. 2003a).

Table 1. Summary of fish monitoring surveys at Te Whanganui a Hei Marine Reserve 1997-2004. BUV = Baited underwater video, UVC = Underwater visual census.

Survey	BUV (no. deployments)	UVC (no. transects)
Autumn 1997	19	198
Spring 1997	35	189
Autumn 1998	36	162
Spring 1998	29	117
Autumn 1999	30	-
Spring 1999	-	180
Autumn 2000	30	165
Spring 2000	30	171
Autumn 2001	26	210
Spring 2001	30	168
Autumn 2002	30	162
Autumn 2003	30	162
Autumn 2004	30	180

Terminology/Abbreviations

ANOVA: analysis of variance.

BUV: baited underwater video, a sampling method developed specifically to survey snapper over small spatial scales. For a full description see Willis & Babcock (2000).

CAP: canonical analysis of principal coordinates, a constrained ordination technique for testing *a priori* hypotheses about multivariate data (see Appendix 1 of Willis et al. 2003a for further details).

CI: confidence interval.

DISTLM: computer program used to run distance-based multivariate analysis for a linear model (Anderson 2004).

FL: fork length.

GPS: global positioning system.

JUVsna: the number of snapper less than the recreational size limit of 270 mm fork length.

LEGsna: the number of snapper larger than the recreational size limit of 270 mm fork length.

MAXsna: the total number of snapper seen in a 30 min BUV sequence.

mMDS: metric multidimensional scaling (= PCO: principal coordinate analysis).

NR: non-reserve.

PERMANOVA: permutational multivariate analysis of variance (formerly NPMANOVA).

PERMDISP: computer programme used to test homogeneity of multivariate dispersions (formerly NPDisp; Anderson 2000).

PCO: principal coordinate analysis, an unconstrained ordination technique for visualising multivariate data in two dimensions (see Appendix 1 of Willis et al. 2003a for further details).

R: reserve.

SE: standard error of the mean.

Status: as a factor in a model, the comparison of reserve versus non-reserve densities.

UVC: underwater visual census, a survey method where scuba divers count fish in 25 m × 5 m transects.

METHODS

Survey design

The 2004 census of the Te Whanganui a Hei Marine Reserve was done from June 1-3. Data for previous years were taken from Taylor et al. (2003a).

The reserve and environs were divided into six survey areas (three reserve and three non-reserve, Fig. 1). Within each area, sites were selected to encompass the variability in habitat types as well as geographic coverage of the areas. Three reef sites per area were selected for underwater visual census (UVC), and five sites per area for baited underwater video (BUV) deployments. Power analysis of data from previous surveys indicated that this level of replication was sufficient to detect effect sizes (in terms of reserve:non-reserve ratio of snapper density) of 2.3 for MAXsna and 5.3 for LEGsna, with power set at 0.8 (Willis et al. 2003b). The BUV deployments were haphazardly distributed, although constrained by bottom topography, weather, and current conditions.

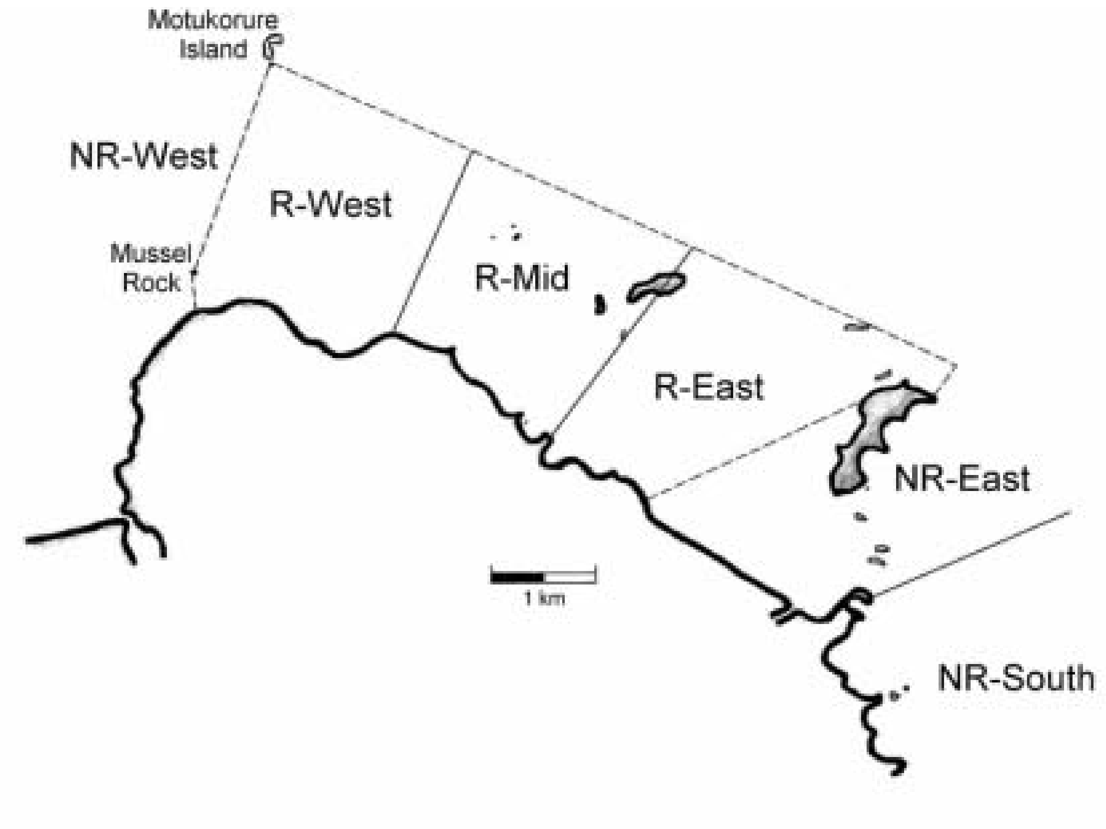

Figure 1. Map of sampling areas in and around the Te Whanganui a Hei Marine Reserve. The dashed line shows the reserve boundary. NR = non-reserve, R = reserve.

Survey methods

Survey methods used here are the same as for previous surveys at Hahei (Taylor et al. 2003a), Leigh (Taylor et al. 2003b), and the Poor Knights Islands (Willis & Denny 2000), except for the use of an additional camera (see next section).

Baited underwater video

BUV sampling was done using two cameras deployed from the University of Auckland's R. V. Hawere. Each camera was mounted on a frame with attached bait holder (Fig. 2). The bait holder contained four pilchards (*Sardinops neopilchardus*) that were broken up to maximise the odour plume, and a fifth whole pilchard was cable-tied to the lid. Fresh baits were used for each replicate. Prior to deployment, location data (including GPS coordinates), depth, and time were written down and filmed so that each video sequence was introduced by this information. The recorder for one of the two camera systems was situated on the anchored Hawere, and connected to the camera by a cable. In the second (new) system, we used a self-contained Sony digital camcorder in an underwater housing, so that it could be dropped and retrieved later via a surface float, with no anchoring of the vessel required. The field of view was the same as for the original BUV system to ensure that results were comparable. The use of a second camera enabled us to reduce field time by running two BUV stations simultaneously. All video sequences were of 30 min duration (from the time the unit contacted the seabed).

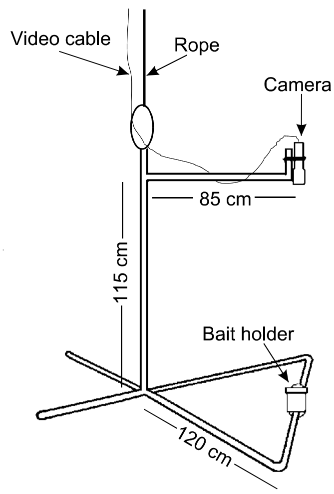

Figure 2. Baited underwater video assembly, with dimensions of the stand.

Analysis of video footage

Videotapes were played back on a VCR with a real-time counter, and the number of each species of fish present at the bait enumerated at 30 s intervals. The maximum number of snapper (MAXsna) and the maximum number of blue cod (MAXcod) present at the bait during each 30 min sequence were recorded, as well as the time from deployment at which each count was made (i. e., t_{MAXsna}, t_{MAXcod}). The MAX index has been previously shown to provide the best estimates of snapper and blue cod relative density (Willis & Babcock 2000, Willis et al. 2000). Individual fish were measured by digitising video images using the SigmaScan® image analysis system, and obtaining a three-point calibration (to compensate for wide-angle distortion) for each image using the marks visible on the base quadrat. Measurements were usually only made of those fish present within the quadrat when the count of the maximum number of fish of a given species in a sequence (e. g., MAXsna) was made. The only exception to this rule was where fish were seen elsewhere in the sequence that were obviously different fish, by virtue of size (i. e., differed from MAXsna measurements by > 100 mm). Small snapper that appeared early in the sequence were the most frequent additions to the dataset, but sometimes one or two large fish were measured in this way. While this meant that some fish moving in and out of the field of view might not have been measured, it also avoided repeated measurement of the same individuals.

The ability to measure fish length allowed the acquisition of three forms of snapper relative density data: the maximum number, and the number of fish > or < minimum legal size (e. g., MAX sna, LEGsna, JUVsna).

Underwater visual census

Within each site, two or three divers surveyed fishes within a total of ten 25 m Δ 5 m transects. A diver fastened a fibreglass tape to the substratum, then swam 5 m before commencing counts to avoid sampling fish attracted to the diver. The tape was swum out to 30 m, with all fish visible 2.5 m either side of the swim direction included. Where certain schooling species (especially sweep *Scorpis lineolatus*) were too numerous to be counted, numbers were estimated in hundreds. Occasionally, blue cod would follow divers between transects, and care was taken not to include these individuals in subsequent transect replicates. Fish species that were observed outside the transects were recorded as present. Depth and broad habitat type were recorded for each transect.

To quantify the gradient in wave exposure across the survey areas, the software Fetch Effect Analysis (by Eduardo Villouta) was used to calculate the total fetch at the centre of the outer boundary of each survey area (in 10° increments to a maximum of 300 km).

Statistical analyses

Univariate analyses

Because the variable of interest is ‘count’ data, rather than measurement of a continuous variable, traditional linear modelling and hypothesis testing (e. g., analysis of variance, ANOVA) may not be appropriate. Count data of organisms are often not normally distributed and also tend to have heterogeneous variances among samples, because the variance is generally a function of the mean (e. g., Taylor 1961). Such data therefore generally violate the assumptions of traditional linear models, resulting in unreliable results. In 2004 densities estimated using BUV data were tested for normality using the Shapiro-Wilks test. Homogeneity of variances was tested using Levene's test. Three univariate variables were of particular interest: the density of snapper (i) of all sizes, (ii) of legal size (> 270 mm fork length) and (iii) juveniles (< 270 mm fork length). The Shapiro-Wilks test statistic indicated a significant departure from normality for each of the three variables ($P < 0.001$ in all cases). In addition, there was significant heterogeneity in the distribution of observations between the reserve and non-reserve samples for legal-sized snapper ($P < 0.001$). Thus, traditional normal-theory tests were not appropriate for these data.

Ratios of densities of snapper between reserve and non-reserve areas were therefore assessed using generalised linear models (McCullagh & Nelder 1989). Count data are best modelled using the Poisson distribution, or more generally, as Poisson with possible overdispersion due to the fact that fish may not behave independently of each other. The log-linear model with correction for overdispersion was fitted using quasi-maximum likelihood with the R statistical package (Ihaka & Gentleman 1996). This expresses the fish counts, Y, as

$$Y \sim Poisson(\varsigma)$$

where $Poisson(\varsigma)$ denotes a (possibly overdispersed) Poisson distribution with expected value of ς, and $\log(\varsigma)$ is modelled as a linear function of the effects. For example, the expected count of fish in replicate j in an area of status i (where i = 1 indicates reserve sites and i = 2 indicates non-reserve sites) is modelled by

$$\log(\varsigma_{ij}) = \sigma + \zeta_i$$

where σ is the overall mean and ζ is the parameter corresponding to the status effect to be estimated. For a log-linear model, the estimates of effects are multiplicative in nature. Thus, the estimate of the effect size is given as a ratio between reserve and non-reserve densities. Thus, an estimated ratio of 1 would indicate no effect, an estimated ratio of 2 would indicate that reserve sites have, on average, two times (×2) the density of snapper observed at non-reserve sites, and so on. In accordance with previous assessments, only changes of 100% or greater were regarded as biologically significant. This conservative approach reduces the probability of committing a Type I error (i. e., rejecting the null hypothesis where in fact no real difference exists).

A two-tailed t-test was used to test for differences in mean sizes of snapper inside versus outside the reserve.

Multivariate analyses

Multispecies UVC data were examined using both univariate and multivariate techniques. All multivariate analyses were done using data pooled at the level of individual stations (i. e., the n = 10 transects were summed for each variable to obtain a single observation for each station). There were 32 fish species variables recorded and included in analyses and a total of 18 multivariate observations, consisting of 3 stations within each of 6 areas, with 3 areas located inside the reserve (areas 2, 3 and

4) and 3 areas located outside the reserve (areas 1, 5 and 6). The pelagic species kahawai, koheru, and jack mackerel were removed for these analyses.

All multivariate methods were based on Bray-Curtis dissimilarities (Bray & Curtis 1957) calculated among observations for data transformed to**Error! Objects cannot be created from editing field codes.**. Whole assemblages were analysed using permutational multivariate analysis of variance (PERMANOVA, Anderson 2001), with “Status” (reserve versus non-reserve) treated as a fixed factor and “Areas” treated as a random factor, nested within “Status”. The factor “Areas” was tested using 9999 random permutations of the raw data. However, with only 3 areas per status category, there were not enough possible permutations to obtain a reasonable test of the factor “Status” using permutation approaches. Thus, an appropriate P-value for this was obtained by randomly drawing a Monte Carlo sample (of size n = 9999) from the asymptotic permutation distribution (as derived by Anderson & Robinson 2003). Data were also tested for homogeneity of multivariate dispersions using the computer programme PERMDISP (Anderson 2000), using these same permutation strategies for the relevant factors. Relative dissimilarities among the fish assemblages observed at different stations were visualized using principal coordinate analysis (PCO, Gower 1966), also known as metric multi-dimensional scaling (mMDS).

The effect of marine reserve status on fish assemblages was also examined using canonical analysis of principal coordinates (CAP, Anderson & Robinson 2003, Anderson & Willis 2003). CAP is a constrained ordination method that is effectively a PCO followed by a traditional canonical discriminant analysis on an appropriate number of the PCO axes. It allows one to find an axis through the multivariate cloud that is best at discriminating group differences, if such differences do indeed exist in the multivariate space. The P-value for the multivariate CAP test was obtained using 9999 permutations of the raw data. For further details concerning any of the multivariate methods used in the present investigation, see Appendix 1 of Willis et al. (2003a).

The effect of the environmental gradient running across the survey areas on the fish assemblage was examined as follows. We assigned values of 1-6 to the areas running from west to east, and fitted this gradient to the UVC data with permutational multivariate multiple regression, using the software DISTLM (Anderson 2004). The reserve effect was then examined after the environmental gradient was used as a covariate (i. e., the environmental gradient was “factored out” or “partitioned out”).

RESULTS

Baited underwater video

Snapper Pagrus auratus

Results from the BUV indicate that snapper of all sizes combined continue to be more abundant inside the marine reserve than outside, by a factor of 3.0 for the June 2004 survey (Table 2, Fig. 3a). As in previous years, this difference was mainly due to legal-sized (> 270 mm FL) snapper, which were estimated to be 18.3 times more abundant inside the reserve than outside (Table 2, Fig. 3b). Numbers of undersize fish differed little with respect to reserve status (Table 2, Fig. 3c). Within the reserve, average numbers of snapper recorded using the BUV were within the range of values recorded during previous years for all snapper and for both size categories, and no long-term trends are evident (Figs 3a-c). The increase in the ratio of legal-sized snapper inside versus outside the reserve from 9.0:1 in 2003 to 18.3:1 in 2004 was due to a decline in fish numbers outside the reserve rather than an increase inside (Fig. 3b). There was no evidence of a peak in relative densities of legal-sized snapper in the centre of the reserve that would be consistent with an effect of strong fishing pressure at the outer boundaries (Fig. 4), although with only three areas inside the reserve our ability to detect such a peak is low. In 2003 densities were relatively constant across the three areas inside the reserve (Fig. 4a), while in 2004 there was a trend for densities to increase from west to east (Fig. 4b).

In June 2004 the average size of all snapper inside the reserve was 89 mm greater than the average for fish outside the reserve, an increase from the value of 69 mm observed in June 2003 but within the range of estimated differences of 30-106 mm from prior surveys (Table 3, Fig. 5). There were insufficient fish outside the reserve to allow for a meaningful comparison of lengths for the legal-sized snapper. In 2004 the average size of all fish within the reserve dropped slightly (by 20 mm) from the high seen in the 2003 survey (Table 3). The size distribution of the reserve snapper population was very similar to that for the 2003 survey, although fewer very large individuals were recorded (Fig. 5).

Table 2. Mean densities of snapper *Pagrus auratus* inside and outside the Te Whanganui a Hei Marine Reserve, from 2000-2004 BUV surveys. Statistically significant (P < 0.05) ratios of reserve (R) to non-reserve (NR) densities are denoted by *, and ← denotes an infinite ratio where the model could not be fitted. MAXsna = all fish, LEGsna = fish > 270 mm fork length, and JUVsna = fish < 270 mm fork length.

Survey	Density measure	Reserve mean	Non-reserve mean	R:NR ratio	Lower 95% CL for ratio	Upper 95% CL for ratio
Spring 2000	MAXsna	2.20	2.53	0.87	0.29	2.06
	LEGsna	0.87	0.07	13.00*	1.61	105.20
	JUVsna	1.33	2.47	0.54	0.16	1.85
Autumn 2001	MAXsna	9.43	7.08	1.33	0.69	2.55
	LEGsna	3.42	0.42	8.23*	1.85	36.55
	JUVsna	6.00	6.67	0.90	0.47	1.72
Spring 2001	MAXsna	5.73	0.60	9.55*	2.51	36.35
	LEGsna	3.07	-	↩*	-	-
	JUVsna	2.67	0.60	4.44*	1.19	16.52
Autumn 2002	MAXsna	7.07	3.20	2.21*	1.14	4.38
	LEGsna	1.87	0.27	7.00*	1.92	25.51
	JUVsna	5.20	2.93	1.77	0.88	3.57
Autumn 2003	MAXsna	6.13	1.67	3.68*	1.34	10.08
	LEGsna	4.80	0.53	9.00*	2.68	30.22
	JUVsna	1.33	1.13	1.18	0.38	3.36
Autumn 2004	MAXsna	5.07	1.67	3.04*	1.11	8.31
	LEGsna	3.67	0.20	18.33*	2.91	115.46
	JUVsna	1.40	1.47	0.95	0.31	3.49

Table 3. Mean sizes of snapper *Pagrus auratus* inside and outside the Te Whanganui a Hei Marine Reserve, from 2000-2004 BUV surveys. Statistically significant ($P < 0.05$) differences are denoted by *. n = number of fish.

Survey	Reserve mean fork length (mm)	n: Reserve	Non-reserve mean fork length (mm)	n: Non-reserve	Difference between means (mm)	95% CI
<u>All snapper</u>						
Spring 2000	249.91	33	143.95	38	105.96*	37.04
Autumn 2001	242.82	137	190.76	85	52.06*	17.08
Spring 2001	277.45	86	208.55	9	68.90*	47.63
Autumn 2002	233.38	106	203.60	48	29.77	19.75
Autumn 2003	323.27	92	254.72	25	68.55*	39.31
Autumn 2004	303.20	76	214.56	25	88.64*	28.79
<u>Legal snapper</u>						
Spring 2000	359.77	13	333.00	1	26.77	156.19
Autumn 2001	320.17	46	288.67	6	31.51	34.98
Spring 2001	328.63	46	-	0	-	-
Autumn 2002	310.86	29	310.50	4	0.36	39.51
Autumn 2003	351.04	72	310.50	8	40.54	63.28
Autumn 2004	329.15	55	322.00	3	7.15	66.38

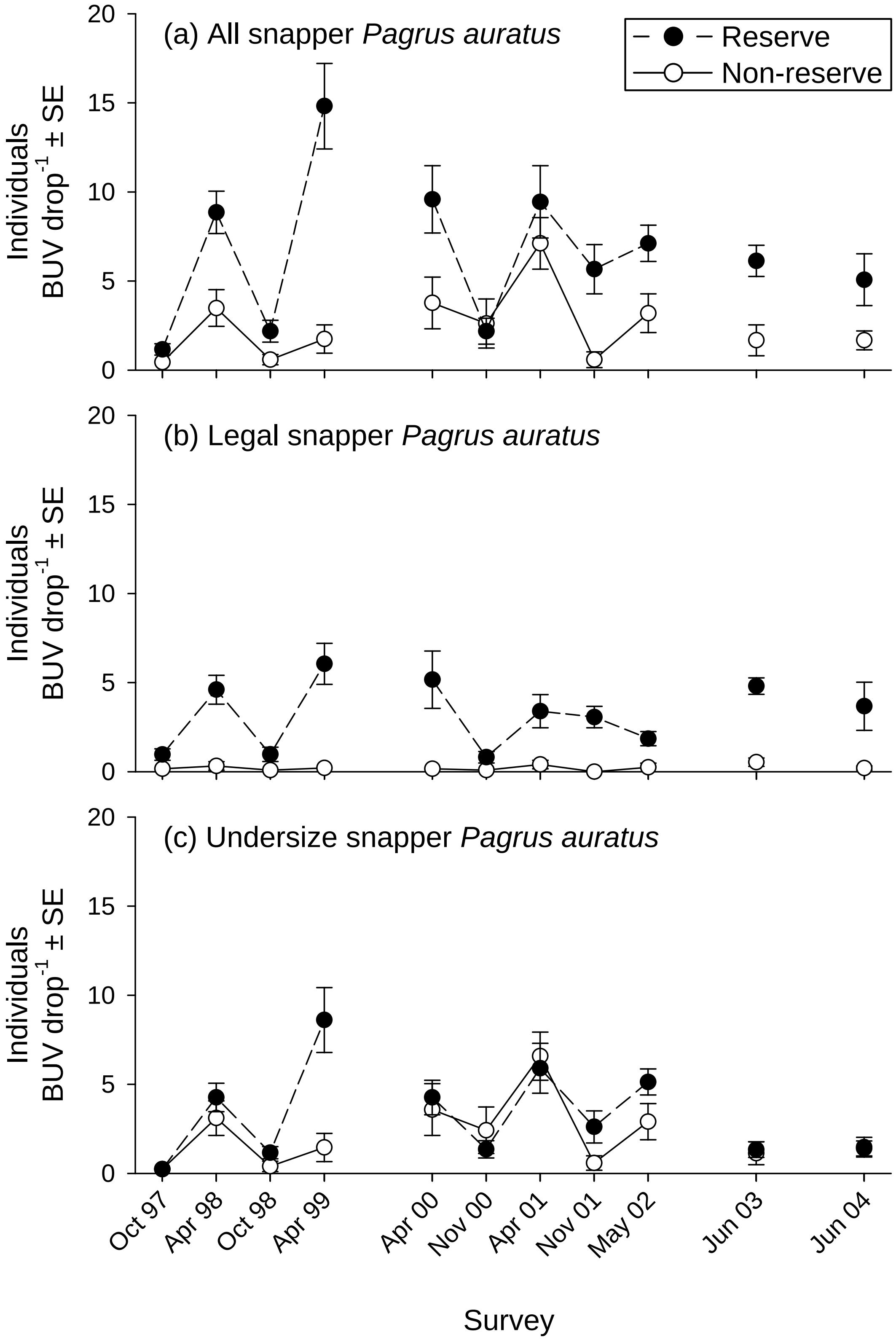

Figure 3. Long-term trends in the relative density of snapper *Pagrus auratus* inside and outside the Te Whanganui a Hei Marine Reserve, as measured using BUV. (a) All snapper (MAXsna), (b) legal snapper (> 270 mm fork length), (c) undersize snapper (< 270 mm fork length).

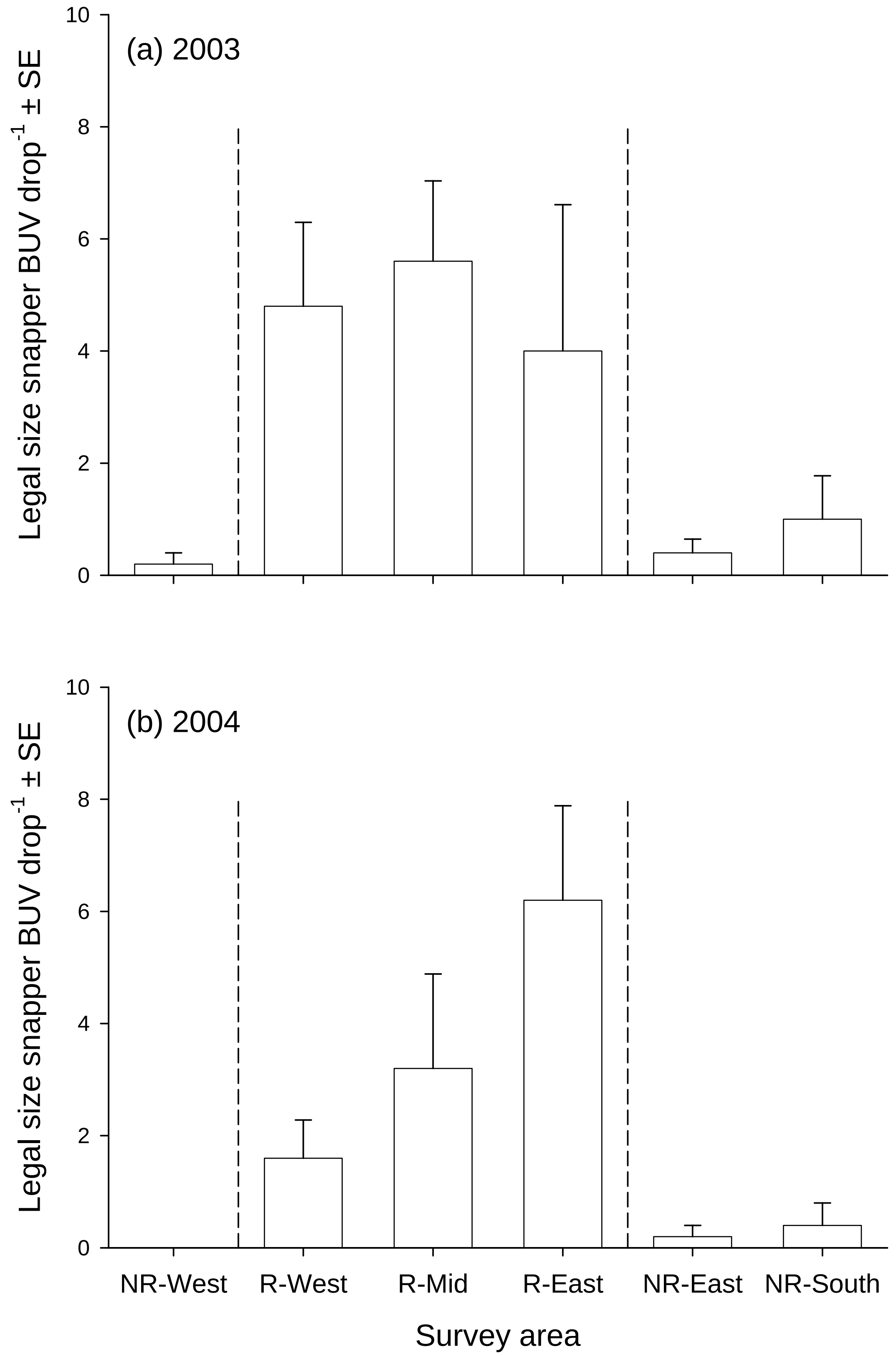

Figure 4. Average number of legal-sized snapper *Pagrus auratus* recorded in the six areas surveyed within and adjacent to the Te Whanganui a Hei Marine Reserve. Dashed vertical lines indicate reserve boundaries.

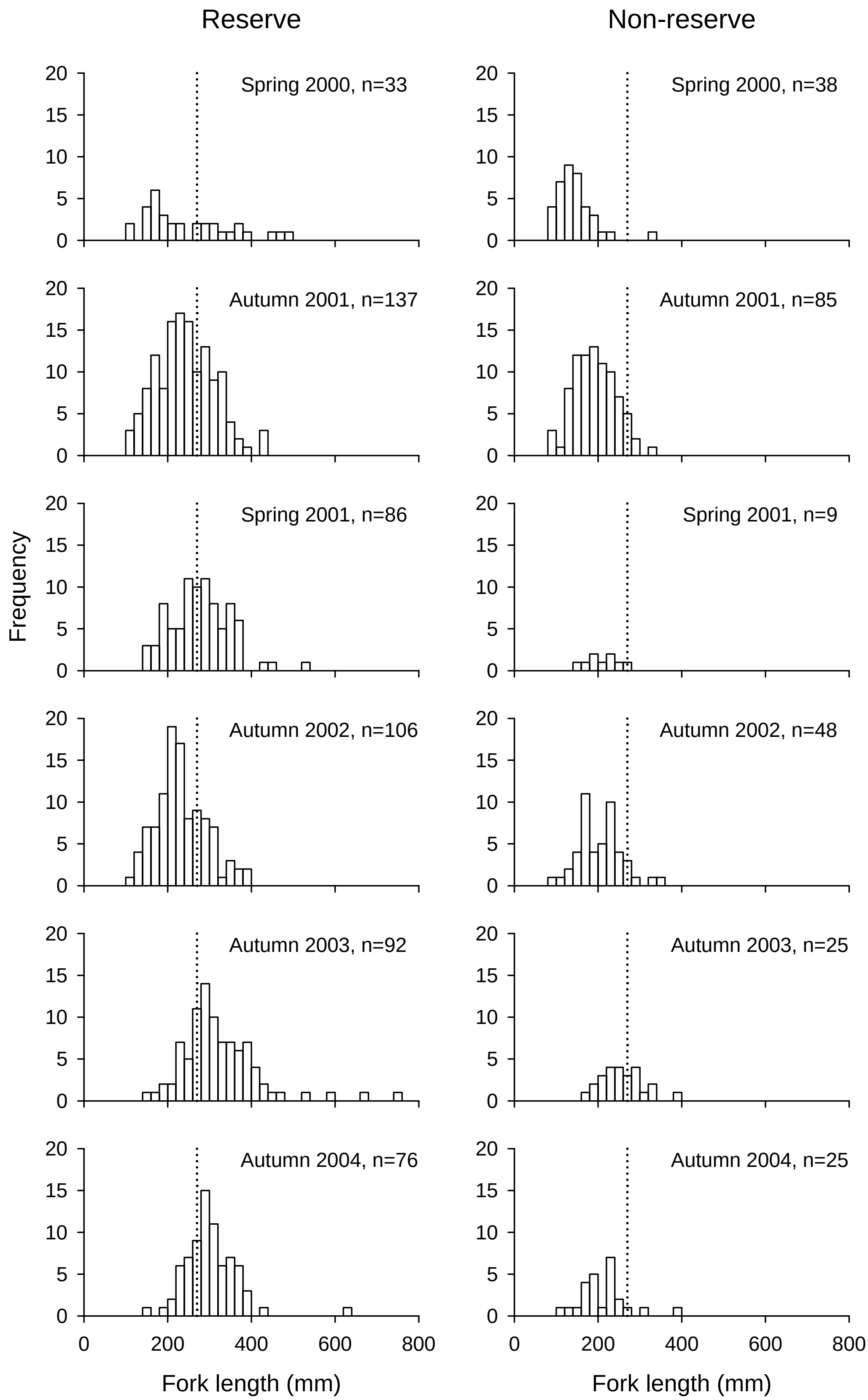

Figure 5. Size frequency distributions of snapper *Pagrus auratus* inside and outside the Te Whanganui a Hei Marine Reserve from 2000-2004, as measured using BUV. Dotted line indicates recreational legal size limit.

Blue cod Parapercis colias

As in previous recent years relatively few blue cod were detected by the BUV in 2004 (Table 4, Fig. 6a). Densities were, on average, 3 times higher inside the reserve than outside (average of 0.60 ± 0.23 (SE) individuals per BUV drop inside versus 0.20 ± 0.12 outside), although this difference was not detected as statistically significant (Table 4). Although there is limited power to detect statistically significant reserve effects for any given year, it is noteworthy that average densities have now been higher inside than outside the reserve in ten of the 11 surveys done since monitoring began in 1997. There is some suggestion that numbers in the reserve have stabilised above the low values recorded in most of the 2000-2002 surveys, but any apparent trends must be interpreted cautiously due to the very low numbers of individual fish recorded during the surveys (e.g., a total of nine from the reserve in 2004). Willis et al. (2003a) suggested that the decline in cod densities might be attributable to warmer than average sea surface temperatures that occurred from the winter of 1998 until the end of 1999 (Fig. 6b), but a much longer time-series would be required to test this hypothesis.

As in previous years a meaningful comparison of sizes of blue cod inside versus outside the reserve was thwarted by the low numbers of individuals recorded (Table 5).

Table 4. Mean densities of blue cod *Parapercis colias* inside and outside the Te Whanganui a Hei Marine Reserve, from 2000-2004 BUV surveys. Statistically significant ($P < 0.05$) ratios of reserve (R) to non-reserve (NR) densities are denoted by *, and ← denotes an infinite ratio where the model could not be fitted.

Survey	Reserve mean	Non-reserve mean	R:NR ratio	Lower 95% CL for ratio	Upper 95% CL for ratio
Spring 2000	0.00	0.13	0	-	-
Autumn 2001	0.21	0.17	1.28	0.29	5.64
Spring 2001	0.47	0.07	7.00*	0.85	57.37
Autumn 2002	0.20	0.07	3.00	0.49	18.27
Autumn 2003	0.47	0.00	←*	-	-
Autumn 2004	0.60	0.20	: 5/77#	0.51	17.56

Table 5. Mean sizes of blue cod *Parapercis colias* inside and outside the Te Whanganui a Hei Marine Reserve, from 2000-2004 BUV surveys. Statistically significant ($P < 0.05$) differences are denoted by *. n = number of fish.

Survey	Reserve mean fork length (mm)	n: Reserve	Non-reserve mean fork length (mm)	n: Non-reserve	Difference between means (mm)
Spring 2000	-	0	370.00	1	-
Autumn 2001	319.33	3	244.00	2	75.33
Spring 2001	248.00	7	166.00	1	82.00
Autumn 2002	164.00	3	170.00	2	-6.00
Autumn 2003	330.86	7	-	0	-
Autumn 2004	289.56	9	159.67	3	129.89*

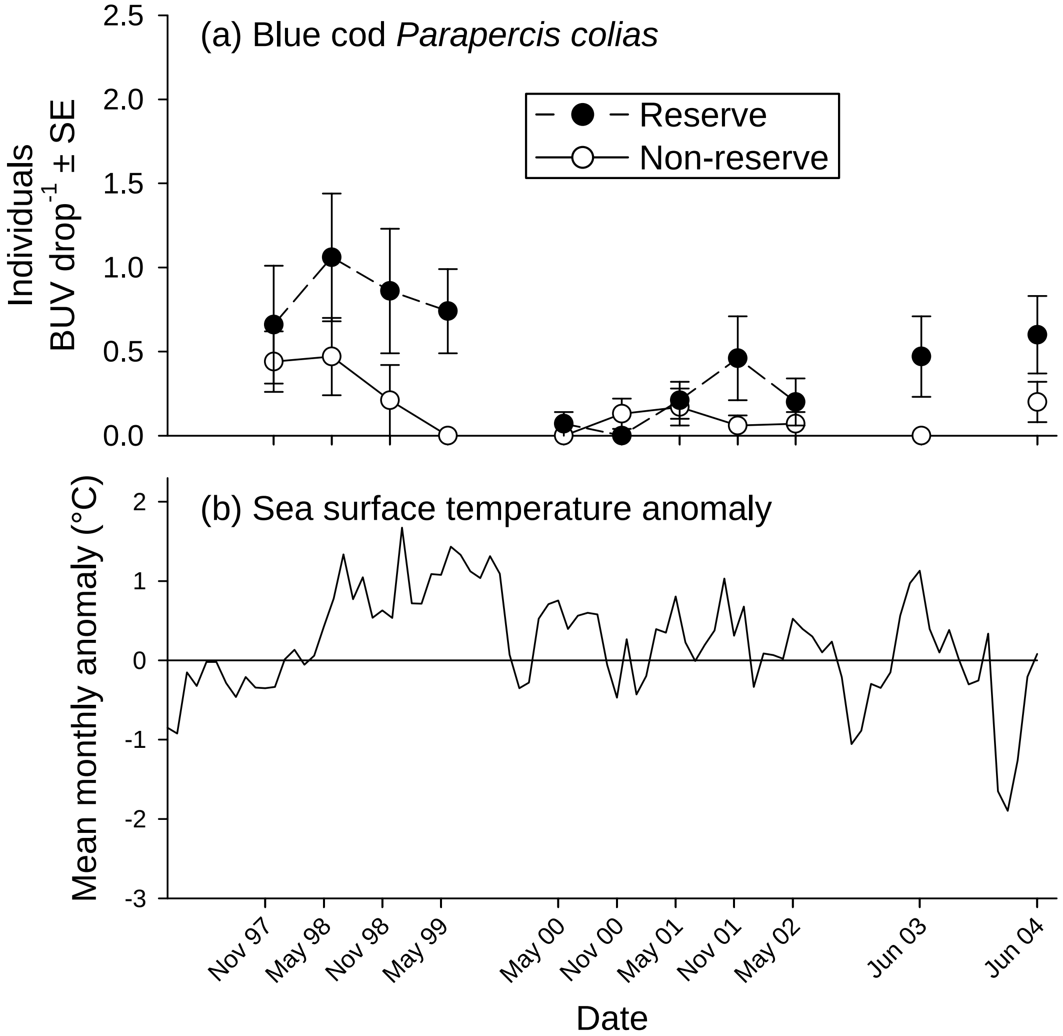

Figure 6. (a) Long-term trends in the density of blue cod *Parapercis colias* inside and outside the Te Whanganui a Hei Point Marine Reserve, as measured using BUV. (b) Leigh sea surface temperature anomalies (from long-term average 1967-96).

Underwater visual census

Community-level patterns

There was no significant difference in the composition of fish assemblages from areas outside the marine reserve compared to those inside the marine reserve (i. e., a non-significant effect of "Status" in the PERMANOVA, Table 6). The average percentage difference among fish assemblages from the non-reserve stations (40.0%) was larger than that from the reserve stations (25.8%). This suggested that assemblages at non-reserve sites might be more variable than those inside the reserve – which is the opposite pattern to that recorded last year. In any case, there was no statistically significant difference in the multivariate dispersion of fish assemblages detected inside versus outside the reserve (PERMDISP, Table 7). Thus, reserve and non-reserve sites were not clearly distinguishable on the basis of either species composition or community variability. In addition, there were no clear or significant differences among individual areas within each status (Tables 6 and 7).

The unconstrained PCO plot suggested, however, that there may be some slight distinction in fish community structure due to reserve status, with assemblages inside the reserve occurring generally in the lower half of the plot (closed symbols), while assemblages outside reserves occurred generally in the upper half of the plot (Fig. 7a). However, in some parts of the diagram, the open and closed symbols were quite well-mixed and thus these two types of assemblages did not separate cleanly in two dimensions on the unconstrained plot. In addition, the variation among individual areas either inside or outside the reserve was just as large as any observed differences between communities from reserve and non-reserve areas (i. e., assemblages from areas inside the reserve tended to be just as far away from each other as assemblages from areas outside the reserve, Fig. 7b). Thus, it is not surprising that the factor "Status" was not statistically significant in the PERMANOVA (Table 6). It is also clear from Fig. 7b that there was no apparent spatial gradient in fish assemblages from one end of the sampling design to the other (i. e., the points are not ordered in the plot from 1 to 6).

In contrast, a significant relationship between the fish assemblage structure and reserve status was apparent in the constrained (CAP) plot (with a squared canonical correlation of 0.725, Fig. 8). There was also some evidence of a reserve effect in assemblage structure using the CAP statistic (P = 0.0595). This multivariate test has greater power than PERMANOVA in the presence of strong correlation structure among variables. The mis-classification error (with a total error = 22%) further indicated that distinguishing these two fish assemblages was possible: the leave-one-out allocation success for reserve sites and for non-reserve sites was 78%. Note that we would expect to get an allocation success of 50% (no better than a random 50-50 chance of being right in the case of two groups) if the null hypothesis of no difference between reserve and non-reserve assemblages were true.

Correlations of individual species with the canonical axis were calculated in order to indicate the species of fish that may be responsible for the differences in assemblage structure that were detected inside versus outside the marine reserve on the basis of the UVC data. These provide only an exploratory indication of which species may be driving the observed multivariate pattern; they do not provide a formal univariate test for these individual species. Species with either positive correlations (indicative of greater frequency or abundance outside reserves) or negative correlations (indicative of greater frequency or abundance inside reserves) are shown in Table 8.

The environmental gradient running across the survey areas from west to east explained 14.3% of the variation in the fish assemblage data (P = 0.0001; Table 9). After the effects of this gradient were partitioned out (removed) by the analysis, reserve status explained only 9.8% of variation in the dataset (P = 0.19).

Table 6. PERMANOVA on the basis of the Bray-Curtis dissimilarity for ln(y+1) transformed species abundance data (32 species). The *P*-value for Areas was obtained using random permutations of the raw data, whereas the *P*-value for Status needed to be obtained using a random Monte Carlo sample from the asymptotic permutation distribution (details provided in Anderson & Robinson 2003).

Source	df	SS	MS	*F*	*P*
Status	1	1767.43	1767.43	1.000	0.4332
Areas(Status)	4	7048.91	1762.23	1.423	0.0960
Residual	12	14857.72	1238.14		
Total	17	23674.06			

Table 7. PERMDISP on the basis of the Bray-Curtis dissimilarity for ln(y+1) transformed species abundance data (32 species). The analysis is essentially an ANOVA on the dissimilarities of individual observations from their area centroids to evaluate relative spread. *P*-values were obtained as described in Table 6.

Source	df	SS	MS	*F*	*P*
Status	1	10.82	10.82	0.102	0.7701
Areas(Status)	4	424.83	106.21	2.052	0.1515
Residual	12	621.02	51.75		
Total	17	1056.67			

Table 8. Species showing either positive or negative correlation with the canonical axis for status, with a positive correlation indicating species associated with reserve areas and a negative correlation indicating species associated with non-reserve areas (only those having values |*r*| > 0.20 and occurring in more than 2 of the stations are listed).

Positive (indicative of non-reserve stations)		***r***
Butterfish	*Odax pullus*	0.567
Black angelfish	*Parma alboscapularis*	0.552
Spotty	*Notolabrus celidotus*	0.343
Blue maomao	*Scorpis violaceus*	0.243
Porae	*Nemadactylus douglasii*	0.206

Negative (indicative of reserve stations)		***r***
Snapper	*Pagrus auratus*	-0.795
Marblefish	*Aplodactylus arctidens*	-0.591
Green wrasse	*Notolabrus inscriptus*	-0.504
Red pigfish	*Bodianus unimaculatus*	-0.419
Scarlet wrasse	*Pseudolabrus miles*	-0.310
Hiwihiwi	*Chironemus marmoratus*	-0.309
Red moki	*Cheilodactylus spectabilis*	-0.298
Yellow moray	*Gymnothorax prasinus*	-0.273
Trevally	*Pseudocaranx dentex*	-0.217

Table 9. Permutational multivariate multiple regression using DISTLM fitting the gradient (values from 1-6, corresponding to areas running from west to east) to the UVC data first, and then the reserve contrast (inside versus outside) after fitting the gradient as a covariate.

Source	df	SS	*F*	*P*	% variance
Gradient	1	3375.47	2.66065	0.0001	14.26
Contrast/gradient	1	2321.24	1.93680	0.1855	9.80
Residual	15	17977.35			
Total	17	23674.06			

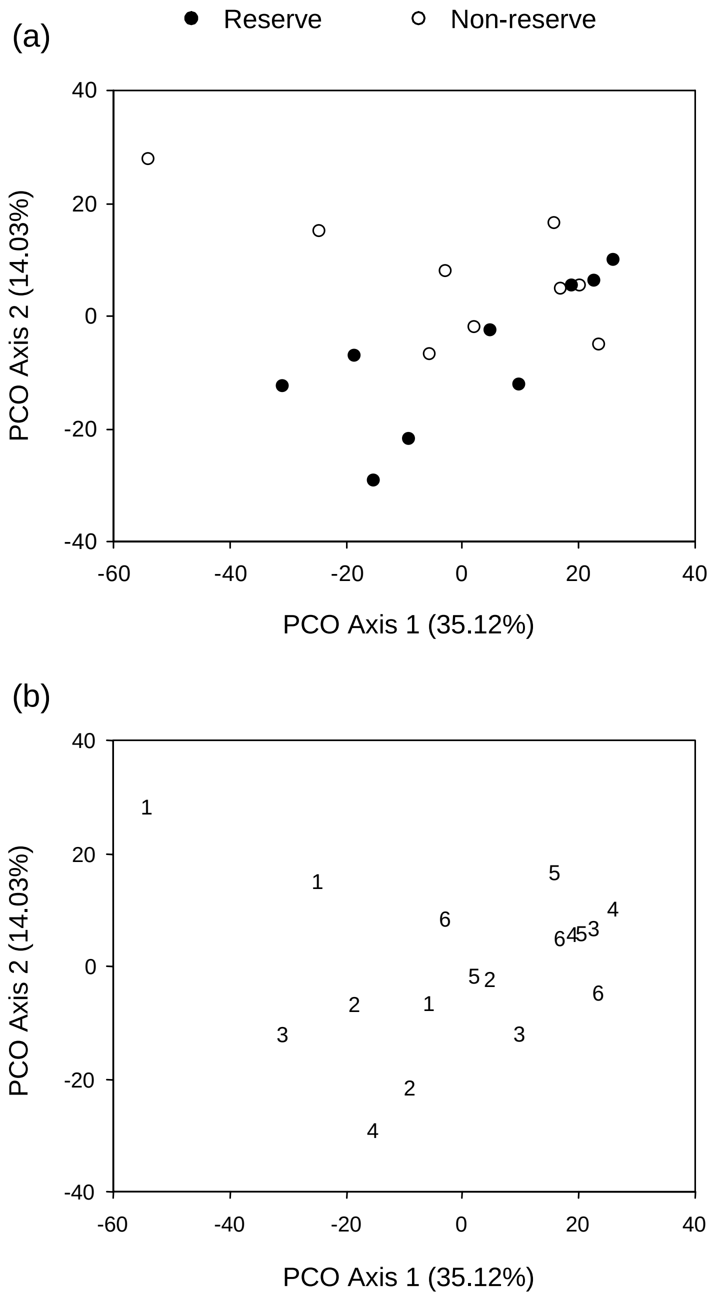

Figure 7. Ordination plot of the first two PCO axes (explaining 49.15% of the original variability) based on Bray-Curtis dissimilarities of ln(y+1) transformed species abundance data (32 species), showing assemblages at different stations with labels for (a) reserve versus non-reserve status or (b) areas 1 through 6 (with 3 stations per area). Areas 1-6 run from west to east, with areas 2, 3, and 4 being inside the reserve and the others outside.

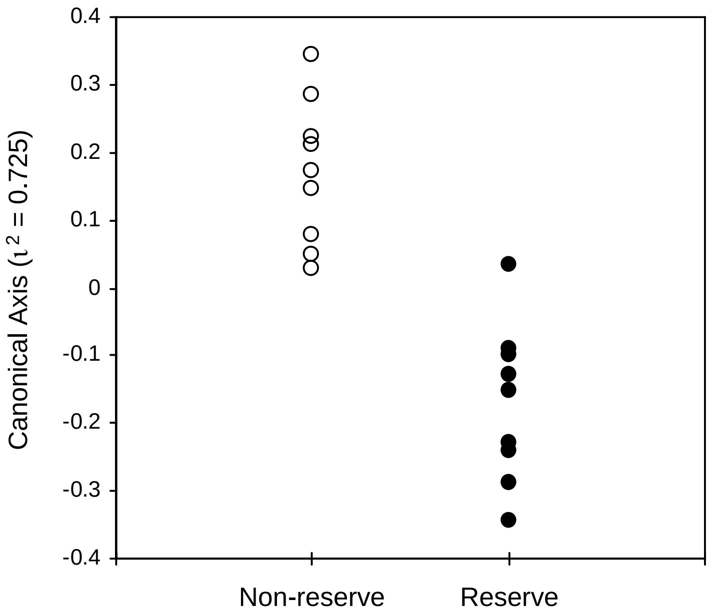

Figure 8. Plot of the canonical axis from a CAP constrained ordination to discriminate fish assemblages from reserve versus non-reserve stations. The discriminant analysis was done on the first m = 8 PCO axes (which explained 98.55% of the original variability) from Bray-Curtis dissimilarities of ln(y+1) transformed species abundances (32 species).

Individual species

Abundances of individual species measured using UVC continued to vary considerably. In 2004, snapper densities inside the reserve returned to levels seen in previous years from the low of 2003, and were 13.3 times higher inside the reserve than outside (Fig. 9). Blue cod densities remained low, and in contrast to the BUV results were similar inside and outside the reserve (Fig. 9). Densities of red moki were similar to those in previous years, with no consistent reserve effect over time (Fig. 9).

Densities of spotties and banded wrasses varied strongly over time, with the populations inside and outside the reserve tracking each other closely, with no apparent reserve effect (Fig. 10). Scarlet wrasse were relatively uncommon in 2003 and 2004, with reserve and non-reserve densities converging, in contrast to previous surveys, which generally found higher densities outside the reserve (Fig. 10).

Abundances of leatherjackets, hiwihiwi, and Sandager's wrasse were similar to 2003 values, with no reserve effect shown for any of these species (Fig. 11).

Densities of goatfish and sweep were similar to 2003 values, with no clear reserve effect for either species (Fig. 12). Large numbers of blue maomao at two sites in the southeasternmost survey area yielded a high mean density for that species outside the reserve (with associated large standard error bars) (Fig. 12).

In 2004 silver drummer and parore (both schooling herbivores) were recorded in very low numbers both inside and outside the reserve (Fig. 13). Demoiselle densities were in line with those recorded prior to the unusually high and variable numbers seen in 2003 (Fig. 13).

Marblefish densities inside the reserve were similar to the previous two years, but no fish were recorded outside the reserve (Fig. 14). No black angelfish were recorded from inside the reserve again in 2004, while numbers outside the reserve remain low (compared to highs in 1997 and 1999) but steady (Fig. 14).

Numbers of species and total numbers of individuals tended to increase from west to east across the study areas (Figs 15a,b). Some habitat parameters potentially responsible for these patterns in the fish assemblage also varied relatively predictably along the coast. Fetch (wave exposure) and transect depth increased from west to east (Figs 15c,d), while the proportion of transects containing significant quantities of sand decreased (Fig. 15e). However, the proportion of transects containing the kelp *Ecklonia radiata* showed no clear pattern (Fig. 15f).

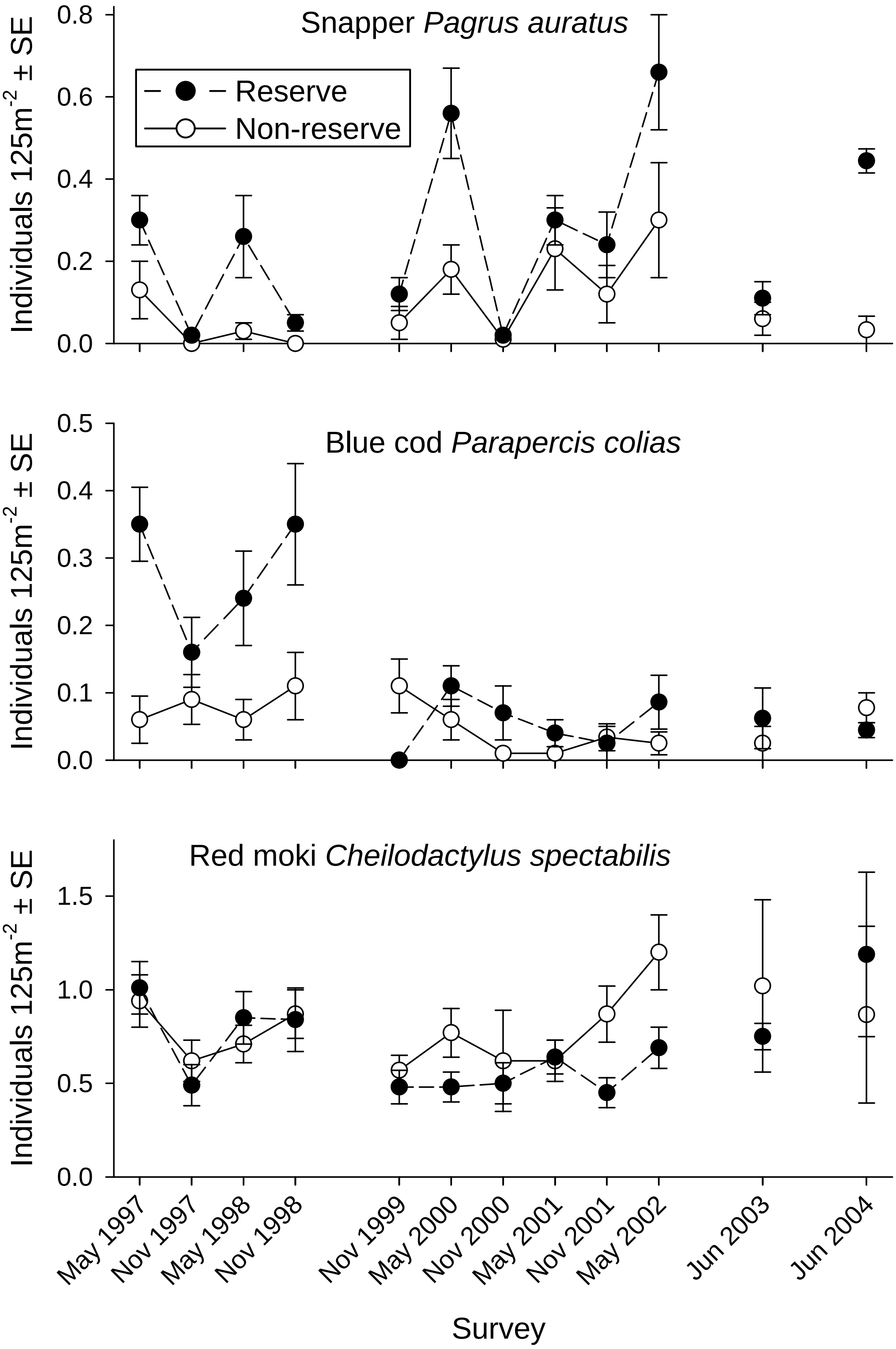

Figure 9. Long-term trends in the densities of snapper, blue cod, and red moki inside and outside the Te Whanganui a Hei Marine Reserve, as measured using UVC.

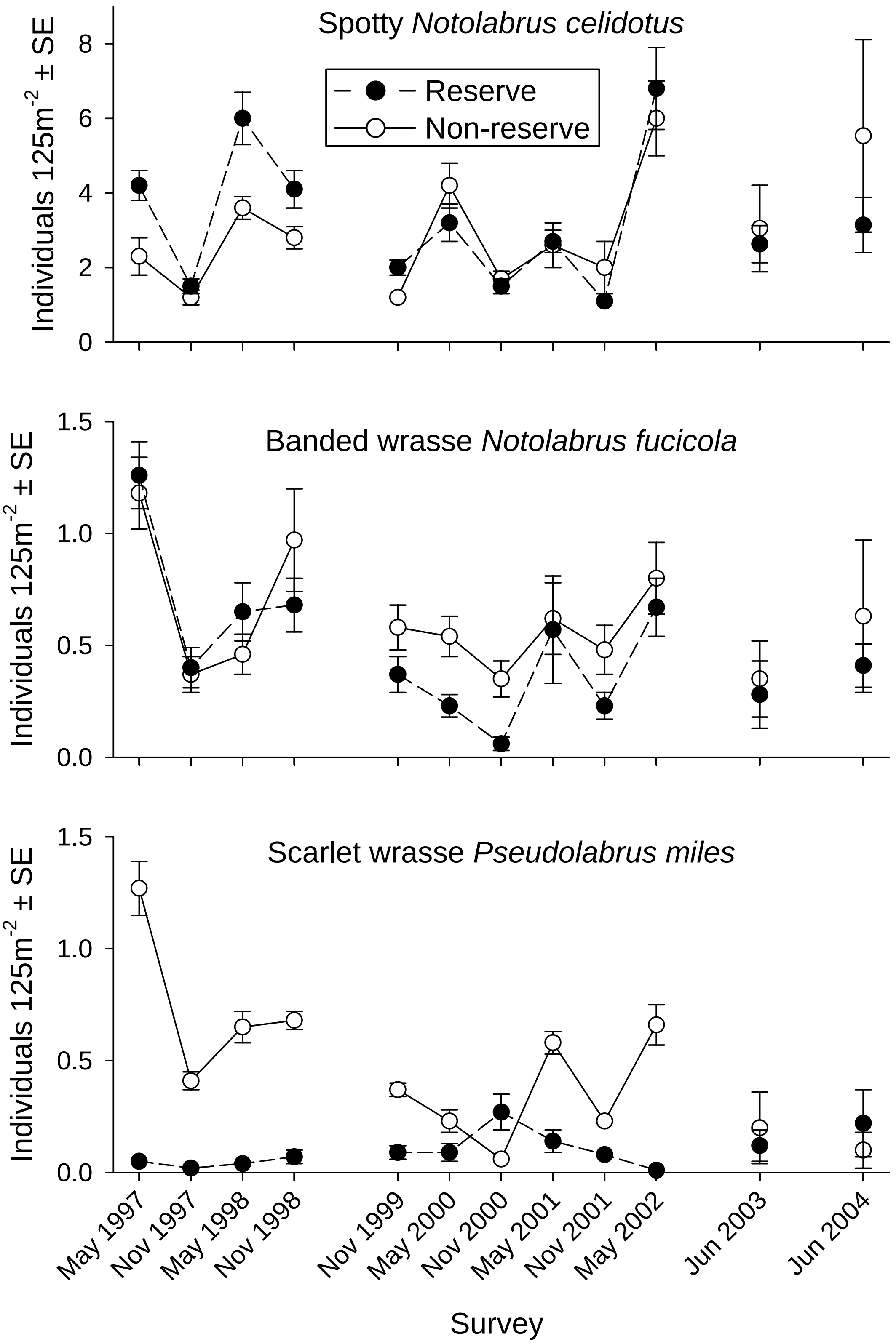

Figure 10. Long-term trends in the densities of spotty, banded wrasse, and scarlet wrasse inside and outside the Te Whanganui a Hei Marine Reserve, as measured using UVC.

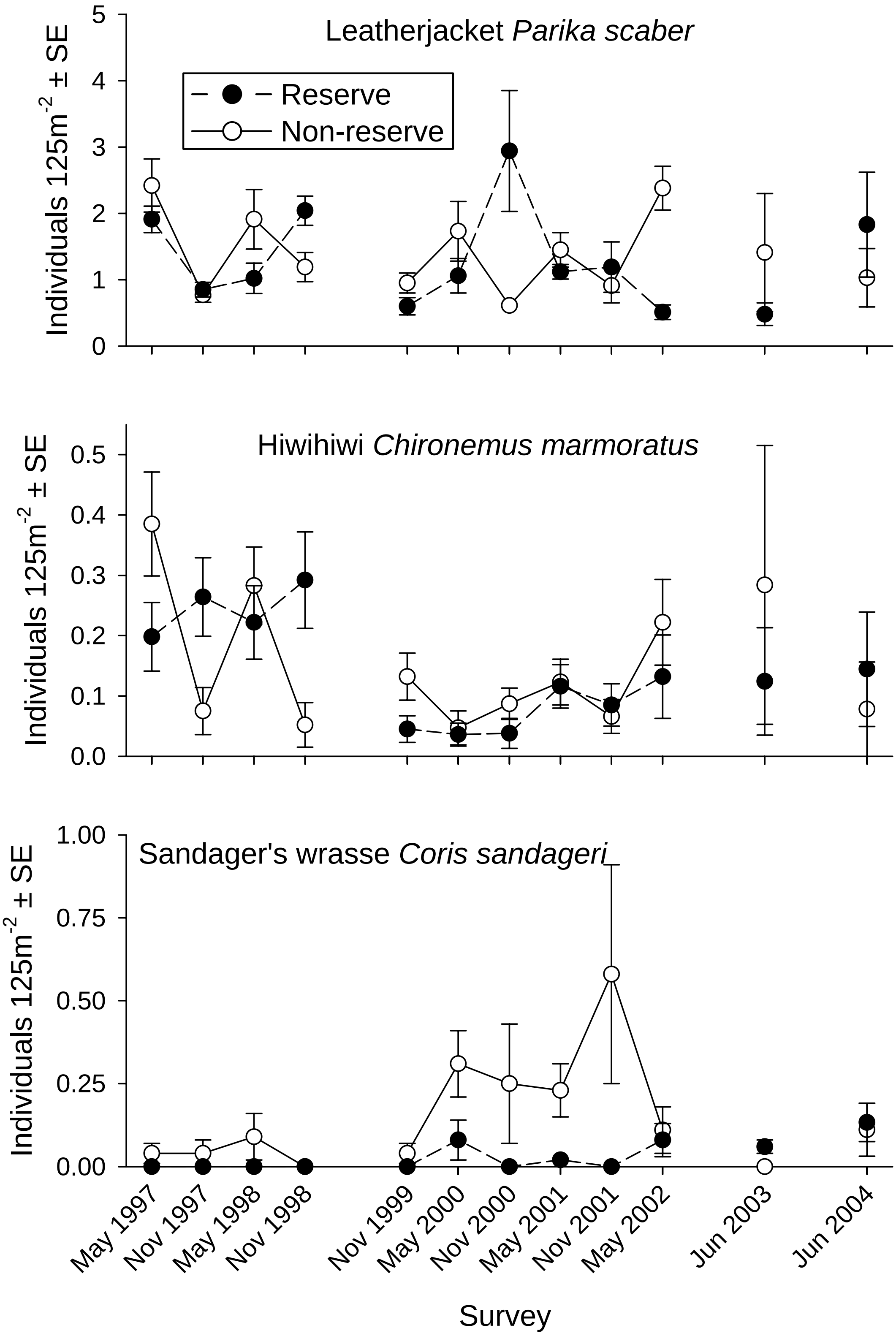

Figure 11. Long-term trends in the densities of leatherjacket, hiwihiwi and Sandager's wrasse inside and outside the Te Whanganui a Hei Marine Reserve, as measured using UVC.

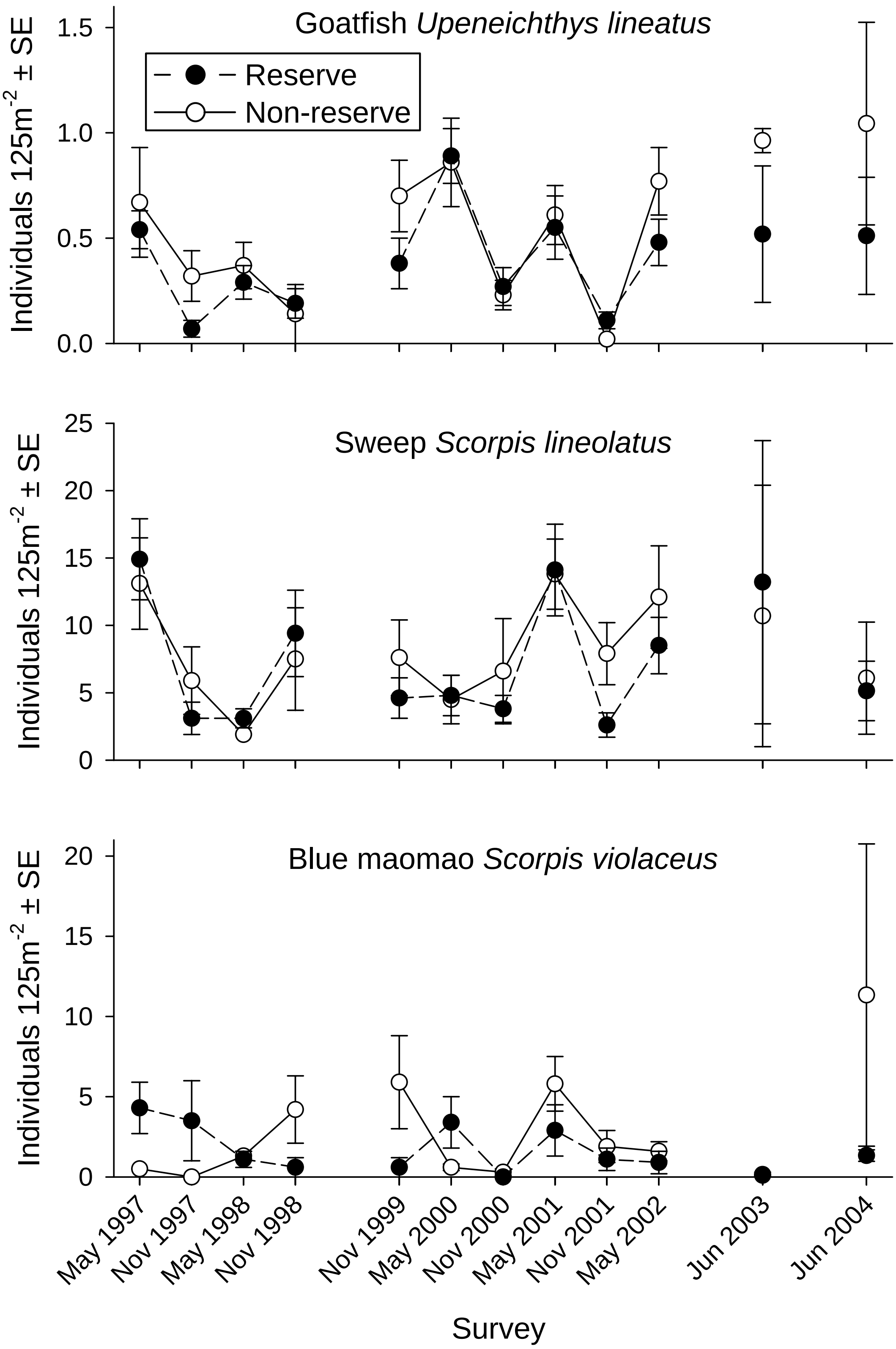

Figure 12. Long-term trends in the densities of goatfish, sweep, and blue maomao inside and outside the Te Whanganui a Hei Marine Reserve, as measured using UVC.

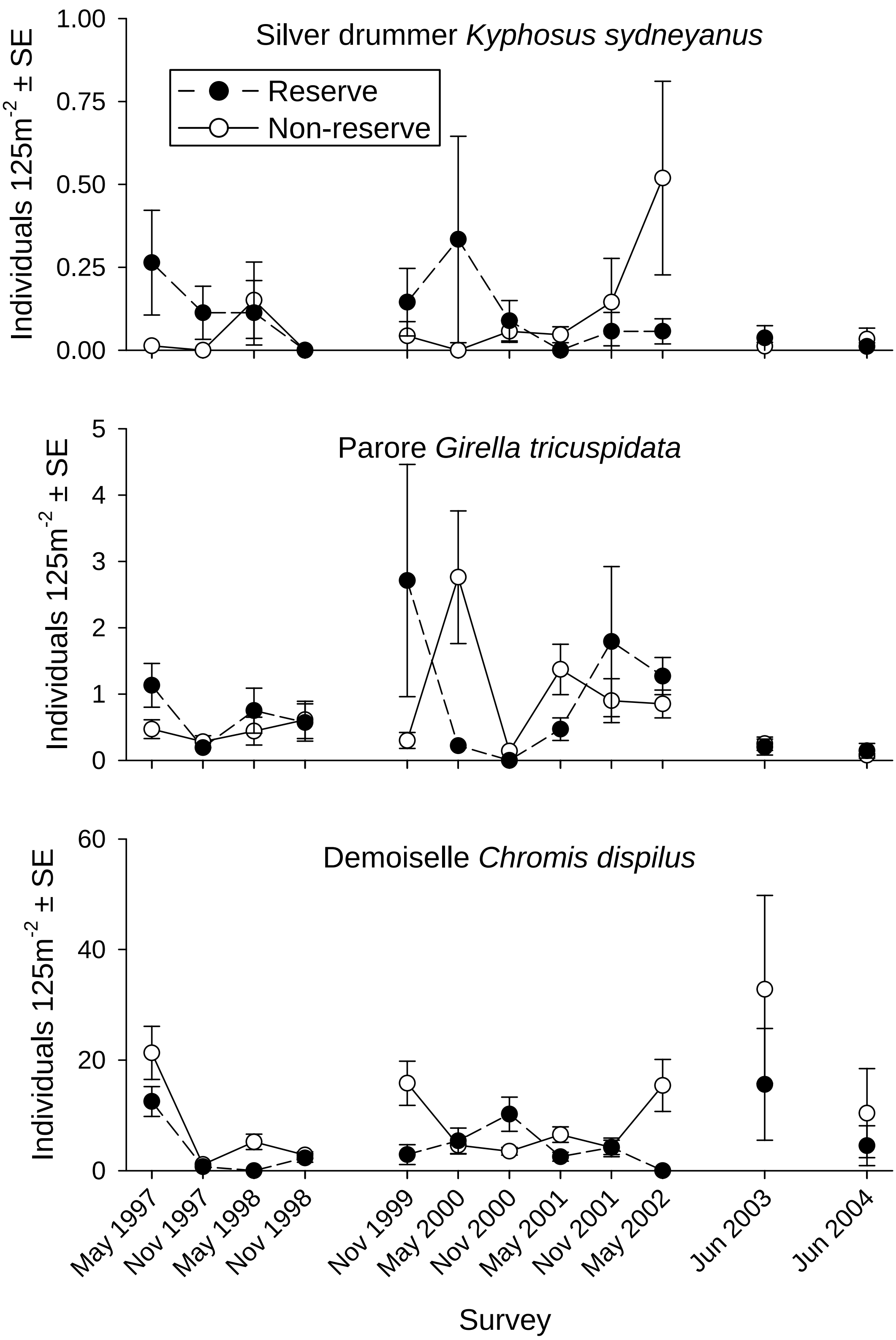

Figure 13. Long-term trends in the densities of silver drummer, parore, and two-spot demoiselle inside and outside the Te Whanganui a Hei Marine Reserve, as measured using UVC.

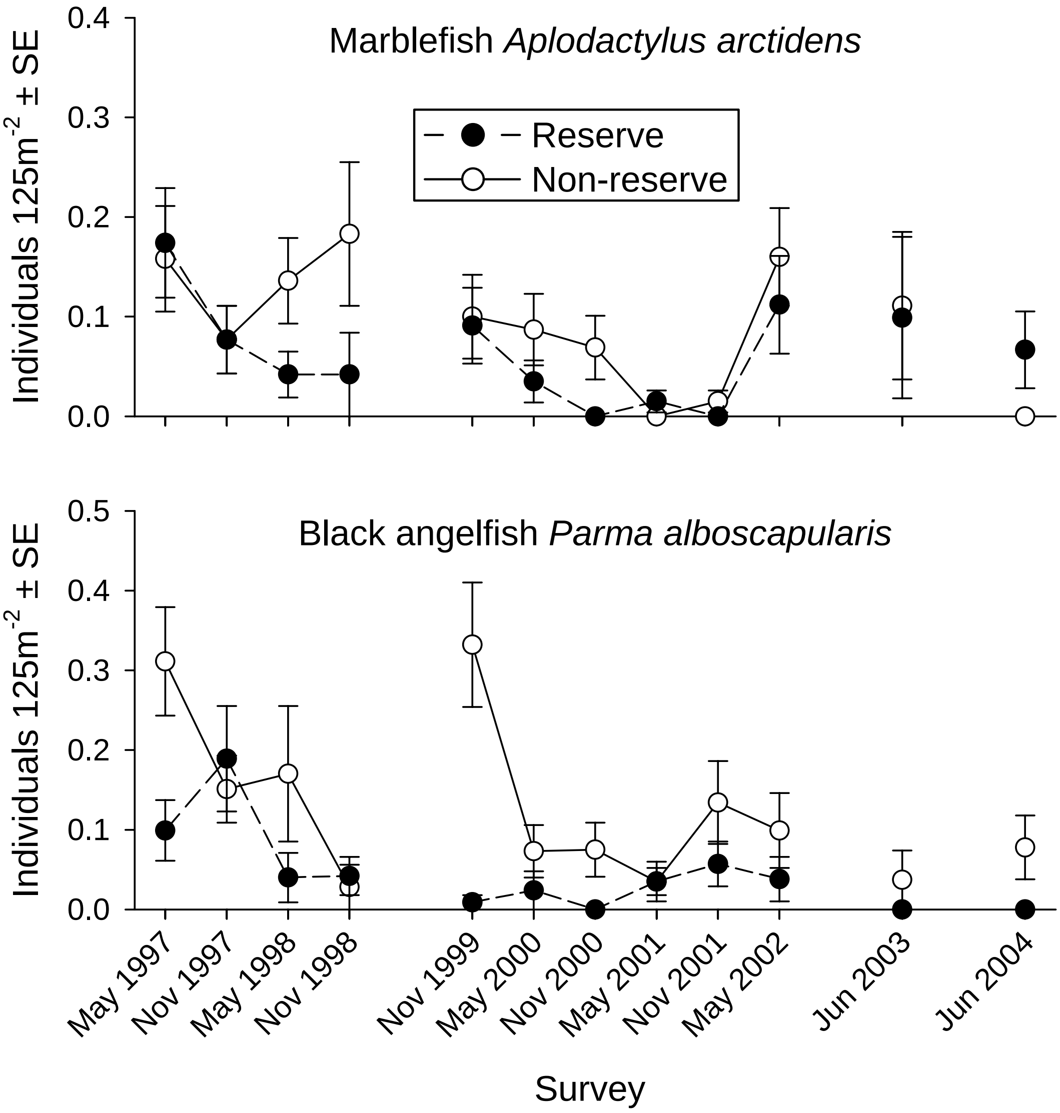

Figure 14. Long-term trends in the densities of marblefish and black angelfish inside and outside the Te Whanganui a Hei Marine Reserve, as measured using UVC.

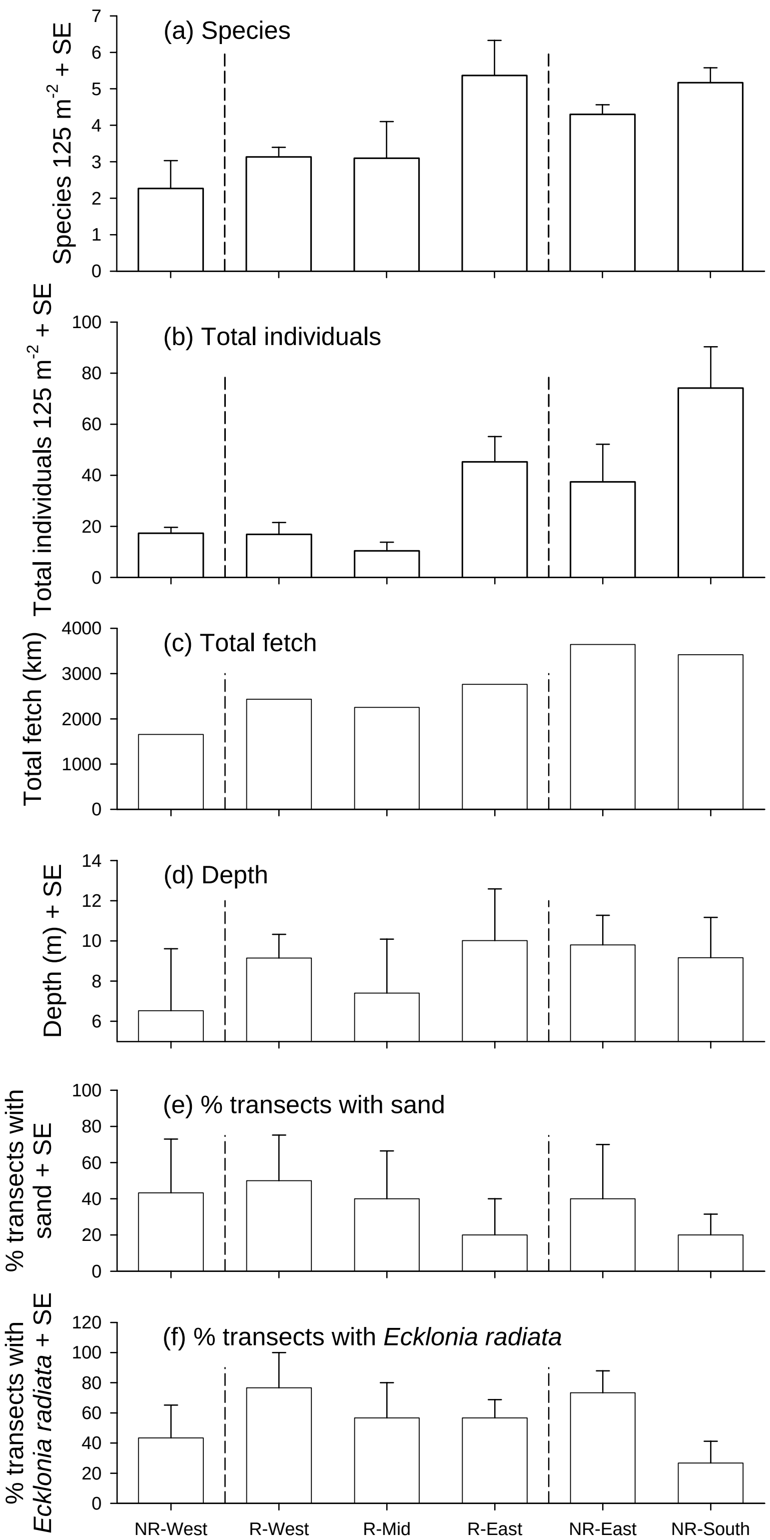

Figure 15. Average number of species and individuals (excluding the schooling pelagic species koheru, kahawai, and jack mackerel) in each survey area inside and outside the Te Whanganui a Hei Marine Reserve, as measured using UVC, plus various physical and biotic habitat descriptors (see text for details regarding fetch calculation). Dashed vertical lines represent reserve boundaries.

DISCUSSION

In line with previous data (Taylor et al. 2003a), the June 2004 baited underwater video (BUV) survey estimated that snapper (*Pagrus auratus*) larger than the minimum recreational legal size of 270 mm fork length were 18 times more abundant inside the reserve than outside. The ratio was higher than for 2003 (9:1) but this change was attributable to a drop in numbers outside the reserve rather than an increase within. Legal-sized snapper are still much less abundant than in the older Cape Rodney to Okakari Point Marine Reserve, and the younger Poor Knight Islands Marine Reserve. There was an average of 2-5 individuals per BUV drop in the Te Whanganui a Hei Marine Reserve during autumn surveys in 2001-2004, compared with 8-22 for the Cape Rodney to Okakari Point Marine Reserve for 2001-2003 (Taylor et al. 2003b) (no survey was done for the latter in 2004), and 8-15 for the Poor Knight Islands Marine Reserve for 2001-2004 (Denny & Shears 2004). Snapper numbers at Hahei appear to be stable, and certainly show no trend to be approaching the densities in the other two reserves. We can only speculate as to why snapper numbers are lower in the Hahei reserve. It is also possible that fishing around the boundaries is depleting snapper numbers throughout the reserve, but if this were occurring we would expect to see higher numbers in the centre of the reserve (as in the Cape Rodney to Okakari Point Marine Reserve; Taylor et al. 2003a), and there is no evidence for such a pattern (Fig. 4; also see Willis et al. 2003b). Another possibility is that environmental conditions are somehow less suitable for snapper – the Hahei reserve appears to be affected more by turbidity and sedimentation than the other two reserves, at least at the western end (R. Taylor, pers. obs.).

As in previous years snapper were significantly larger inside the reserve than outside (by an average of 89 mm FL in 2004). Within the reserve the average size of all fish dropped slightly (by 20 mm) from the high seen in the 2003 survey. This may have been due to the lower numbers of very large fish, as the overall size distribution of the reserve snapper population was similar to that for the 2003 survey.

Blue cod (*Parapercis colias*) densities continue to be higher inside the reserve (though not significantly so for 2004 in isolation), but have not regained the peak densities observed in 1998.

Analyses of the multispecies data gathered using underwater visual census (UVC) revealed some subtle differences in fish assemblages inside versus outside the reserve. The canonical analysis revealed that there was a tendency for communities inside the reserve to contain greater numbers (or frequencies) of snapper, marblefish, pigfish, scarlet wrasse, hiwihiwi, red moki, yellow moray and trevally. On the other hand, greater average numbers (or frequencies) of butterfish, black angelfish, spotties, blue maomao and porae were observed outside the reserve. These patterns were not apparent from multivariate analyses done in previous years and were also not strong compared to the considerable variability in assemblage structure among different areas. In addition, these community trends were not convincing enough to consider a predictive framework – with only a 78% success rate in the classification of fish communities to stations of a particular status.

It is possible that initial effects of reserves are caused by changes in abundances of commercially-targeted species only, with changes in overall fish community structure occurring only after longer periods of time, due to indirect effects of changes in habitat (e. g., Shears & Babcock 2002) or competitive interactions inside the reserve. The fact that a weak community effect was detected this year, in contrast to 2003 but in common with 2002 analyses (when snapper were the only species responding positively to reserve status), indicates that further continued monitoring of fish communities at the assemblage level is warranted.

Given that densities of snapper and blue cod have been relatively stable over the past few years, and that no rapid changes are occurring in the rest of the fish assemblage, the Department of Conservation may wish to consider reducing the frequency of surveys (perhaps to every second year). If this is done surveys should still be carried out at the same time of year with similar levels of replication to ensure comparisons can continue to be made among years.

As noted in the previous reports (Taylor et al. 2003a, Willis et al. 2003a) there is a strong environmental gradient running through the Te Whanganui a Hei Marine Reserve and adjacent reference areas. The western end is shallower, has less reef, and is subject to higher sedimentation and freshwater input (from the Whitianga River), while the eastern end contains a large amount of complex reef and is exposed to greater wave action and clearer, faster-flowing water (Shears et al. 2000). This gradient appears to have an effect on reef fish in the area, with the number of species and individuals tending to increase from west to east without any apparent regard to reserve status (Fig. 15a,b). In 2004 the gradient explained 14.3% of variation in the reef fish assemblage as a whole, which was more than the 9.8% of variation explained by reserve status alone after the effects of the environmental gradient were removed (Table 9). A better understanding of the underlying fish-habitat relationships would enhance our ability to detect the effects of reserve protection on those species responding more subtly than snapper and blue cod.

This would require fine-scale mapping of benthic habitats (probably using a combination of high-frequency sidescan sonar and drop-video), and monitoring of the major habitat-forming organisms (seaweeds, sea urchins, larger gastropods, larger sessile organisms) in the six areas currently surveyed for fishes. In addition to clarifying the effects of reserve protection on fishes, regular monitoring of habitat and benthic organisms would allow for detection of ecosystem-level changes due to, for example, increased sedimentation, or trophic cascades resulting from the effects of higher numbers of predatory fishes and lobsters on sea urchins and in turn seaweeds (Shears & Babcock 2002).

Recommendations

- The fish monitoring programme should be continued with the current levels of sample replication regarded as a minimum level of effort, but the Department of Conservation may wish to consider reducing the frequency of surveys (perhaps to every second year) given that densities of snapper and blue cod have been relatively stable over the past few years, and that no rapid changes are occurring in the rest of the fish assemblage. If this is done surveys should still be carried out at the same time of year to ensure comparability among years.

- The Department should consider initiating the mapping of benthic habitats and the monitoring of important benthic organisms inside and outside the reserve. This would allow for detection of changes in ecosystem-level changes due to, for example, increased sedimentation, or trophic cascades resulting from the effects of higher numbers of predatory fishes and lobsters on sea urchins and in turn seaweeds. Habitat change may also help to explain temporal changes in abundances of many of the fish species currently monitored.

ACKNOWLEDGEMENTS

We thank Murray Birch, skipper of Hawere, and divers Charles Bedford, Pam Brown, Emma Newcombe and Tony Wood for their help in the field. We are also grateful to Arthur Cozens for helping with the logistics, and Jo Evans for continuing to record and compile Leigh marine climate data. Eduardo Villouta generously allowed access to his Fetch Effect Analysis software. Nick Shears made the fetch estimates used in Fig. 15c, and provided useful discussion during the write-up phase.

REFERENCES

Anderson MJ (2000) NPDisp: a FORTRAN computer programme for non-parametric tests of homogeneity of multivariate dispersions (for any 2-factor ANOVA design) using permutations. Department of Statistics, University of Auckland.

Anderson MJ (2001) A new method for non-parametric multivariate analysis of variance. *Austral Ecology* 26: 32-46.

Anderson MJ (2004) DISTLM v. 4: a FORTRAN computer program to calculate a distance-based multivariate analysis for a linear model. Department of Statistics, University of Auckland.

Anderson MJ, Robinson J (2003) Generalized discriminant analysis based on distances. *Australian and New Zealand Journal of Statistics* 45: 301-318.

Anderson MJ, Willis TJ (2003) Canonical analysis of principal coordinates: a useful method of constrained ordination for ecology. *Ecology* 84: 511-525.

Bray JR, Curtis JT (1957) An ordination of the upland forest communities of southern Wisconsin. *Ecological Monographs* 27: 325-349.

Denny CM, Shears NT (2004) Effects of five years of no-take marine reserve protection on reef fish populations at the Poor Knights Islands. *Report to the Department of Conservation*.

Gower JC (1966) Some distance properties of latent root and vector methods used in multivariate analysis. *Biometrika* 53: 325-38.

Ihaka R, Gentleman R (1996) R: A language for data analysis and graphics. *Journal of Computational and Graphical Statistics* 5: 299-314.

McCullagh P, Nelder JA (1989) Generalized linear models. 2nd Edition, Chapman & Hall, London.

Shears N, Walker J, Babcock R (2000) Te Whanganui a Hei Marine Reserve benthic community monitoring programme – 1999/2000. *Report to the Department of Conservation.* 39 p.

Shears NT, Babcock RC (2002) Marine reserves demonstrate top-down control of community structure on temperate reefs. *Oecologia* 132: 131-142.

Taylor LR (1961) Aggregation, variance and the mean. *Nature* 189: 732-735.

Taylor RB, Anderson MJ, Usmar N, Willis TJ (2003a) Te Whanganui a Hei Marine Reserve fish monitoring 2003: final report. *Report to the Department of Conservation*. 34 p.

Taylor RB, Anderson MJ, Egli D, Willis TJ (2003b) Cape Rodney to Okakari Point Marine Reserve fish monitoring 2003: final report. *Report to the Department of Conservation*. 38 p.

Willis TJ, Anderson MJ, Babcock RC (2003a) Te Whanganui a Hei Marine Marine Reserve fish monitoring 2000-2002: final report. *Report to the Department of Conservation*. 42 p.

Willis TJ, Babcock RC (2000) A baited underwater video system for the determination of relative density of carnivorous reef fish. *Marine and Freshwater Research* 51: 755-763.

Willis TJ, Denny CM (2000) Effects of the Poor Knights Islands Marine Reserve on demersal fish populations. *Report to the Department of Conservation, Science and Research, Research Grant 2519*. 57 p.

Willis TJ, Millar RB, Babcock RC (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 TJ, Millar RB, Babcock RC (2003b) Protection of exploited fish in temperate regions: high density and biomass of snapper *Pagrus auratus* (Sparidae) in northern New Zealand marine reserves. *Journal of Applied Ecology* 40: 214-227.