CAPE RODNEY TO OKAKARI POINT MARINE RESERVE FISH MONITORING 2005: FINAL REPORT

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Date: June 2005

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EXECUTIVE SUMMARY

- This report describes the results of a survey of fish abundances in the Cape Rodney to Okakari Point Marine Reserve, northeastern New Zealand. The survey was undertaken in autumn 2005 and continues a time-series that started in 1997.
- The reef fish assemblage in the reserve continues to be distinct from that found in adjacent fished areas. In 2005 the species most indicative of marine reserve areas were snapper *Pagrus auratus*, blue cod *Parapercis colias*, silver drummer *Kyphosus sydneyanus*, and butterfish *Odax pullus*. Other species, notably spotty *Notolabrus celidotus* and hiwihiwi *Chironemus marmoratus*, were more abundant outside the reserve. This probably reflects their affinity for the urchin barrens habitat, which has largely disappeared from the reserve due to predation on sea urchins by snapper and spiny lobster.
- In autumn 2005, estimates made using Baited Underwater Video indicated that legal-sized snapper were 12.8 times more abundant inside the reserve than outside. Densities were slightly lower than the record high measured in 2003. Fluctuations in snapper density most likely reflect seasonal and inter-annual variability in rates of immigration and emigration, superimposed on a relatively stable resident population. The average size of snapper within the reserve has increased steadily by 20-23 mm fork length per year since 2001.
- As in previous years the absence of a plateau in densities of legal snapper in the central reserve may indicate that the majority of the reserve is affected by removal of individuals through fishing outside the boundaries.

Recommendations

- The fish monitoring programme should be continued at one to two year intervals with the current levels of sample replication regarded as a minimum level of effort.
- The programme should be extended to include comparison with projected new reserves (e. g., Great Barrier Island) using identical sampling design and methodology. Comparison of this established reserve with a new reserve will help elucidate the effects of protection on species that are not targeted by fishers.
- Such studies can only be achieved with a long-term commitment to monitoring. Any attempt to monitor new reserves should begin at least two years prior to reserve implementation and continue for at least five years afterward. The programme can then be reviewed based on (1) any changes observed, (2) the rate of such changes, and (3) the degree of seasonal and annual variability observed.
- The increasing number of surveys likely to be needed in an expanded network of marine reserves in New Zealand will require a more consistent and long-

term approach to funding monitoring at regional and national scales, as well as the methodology and personnel to conduct it. Inconsistencies in methods and approach at different reserves would make the results difficult, if not impossible, to compare. Failure to address these issues will compromise the effectiveness of marine reserve monitoring nationwide.

INTRODUCTION

The Cape Rodney to Okakari Point (or Leigh) Marine Reserve was gazetted in 1975, although it only really became established in 1977. It is the oldest no-take marine reserve in New Zealand. A program of regular monitoring of the abundance of reef fishes at this reserve began in 2000 (Willis & Babcock 2000a), although the relative abundance of exploited species (specifically snapper *Pagrus auratus* and blue cod *Parapercis colias*) have been monitored since 1997 (Willis et al. 2003a, Taylor et al. 2003). Prior to this the only studies specifically aimed at estimating reserve effects at Leigh have been those by McCormick & Choat (1987) on red moki *Cheilodactylus spectabilis*, and Cole et al. (1990) who examined a variety of fish species as well as rock lobster *Jasus edwardsii*. The latter study drew on unpublished data collected by A.M. Ayling between 1976-82.

The monitoring of marine reserves has three related, but distinctive functions. First, long-term monitoring datasets can be used to determine whether populations have recovered within reserves relative to fished areas. Second, they allow an assessment of the natural variability associated with species abundance in particular locations, and therefore can detect if changes occur in the biota. These might come about either as a result of sudden (pulse) disturbances, or as gradual (press) changes that may or may not be of natural origin. Third, long-term monitoring data assist in the interpretation of environmental and habitat changes arising indirectly from changes in the relative density of predators (trophic cascades).

In the absence of comparable data collected prior to reserve establishment, comparison of trends in fish numbers inside and outside of several reserves is our best opportunity to determine recolonisation rates of depleted fish species to protected areas. Surveys at Leigh have been run concurrently with surveys at the Te Whanganui a Hei Marine Reserve (Willis 2000), with a view to making such comparisons.

Fish surveys at Leigh from 2000-2005 were done using two separate, but concurrently run methodologies. Carnivorous fishes, which are commonly exploited by fishers, were surveyed using baited underwater video (BUV: Willis & Babcock 2000b, Willis et al. 2000). This method allows the collection of both relative density and size data from species (especially the snapper *Pagrus auratus*) that are not amenable to sampling using traditional diver census methods (e. g., Cole 1994, Willis & Babcock 2000b, Willis et al. 2000). The remainder of the demersal reef species were surveyed using underwater visual census (UVC) transects.

Previous BUV surveys at Leigh from 1997-2003 found 7-90 times more legal-size snapper (> 270 mm fork length) inside the reserve than outside (Taylor et al. 2003), with marked seasonal variation in abundance. After several years in which average autumn densities in the reserve were relatively constant (ranging from 12-15 snapper/BUV drop), there was a large increase to 27 fish/BUV drop in 2003. This appeared to be due to a particularly large seasonal influx of snapper during the previous summer, but it was unclear whether the increased densities would be sustained over subsequent years.

This report presents the results of a survey conducted during autumn 2005 using identical techniques to previous years. This report should be read in conjunction with the previous reports (Willis & Babcock 2000a, Willis et al. 2003b, Taylor et al. 2003).

Terminology/Abbreviations

In this report, we use the following terminology and abbreviations:

ANOVA: analysis of variance.

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

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. 2003b for further details).

GLM: generalised linear models.

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).

nMDS: non-metric multidimensional scaling. An unconstrained ordination technique for visualising multivariate data in two dimensions (see Appendix 1 of Willis et al. 2003b for further explanation).

PCO: principal coordinate analysis. An unconstrained ordination technique for visualising multivariate data in two dimensions (see Appendix 1 of Willis et al. 2003b for further explanation).

PERMANOVA: permutational multivariate analysis of variance (Anderson 2001a).

PERMDISP: permutational analysis of multivariate dispersions (Anderson 2004).

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

UVC: underwater visual census. Sampling method utilising scuba divers to count fish in 25 m × 5 m transects.

METHODS

Survey design

The 2005 census of the Cape Rodney to Okakari Point Marine Reserve was done from April 26-May 2 (BUV) and April 26-28 (UVC). Data for previous years were taken from Taylor et al. (2003).

The survey design and methods were identical to those used by Willis et al. (2003b) in past surveys. Survey sites were selected following a randomised block design. The reserve and environs were divided into twelve survey areas (six reserve and six 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. Two reef sites per area were selected for underwater visual census, and four sites per area for video 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. 2003a). The BUV deployments were haphazardly distributed, although constrained by bottom topography, weather, and current conditions.

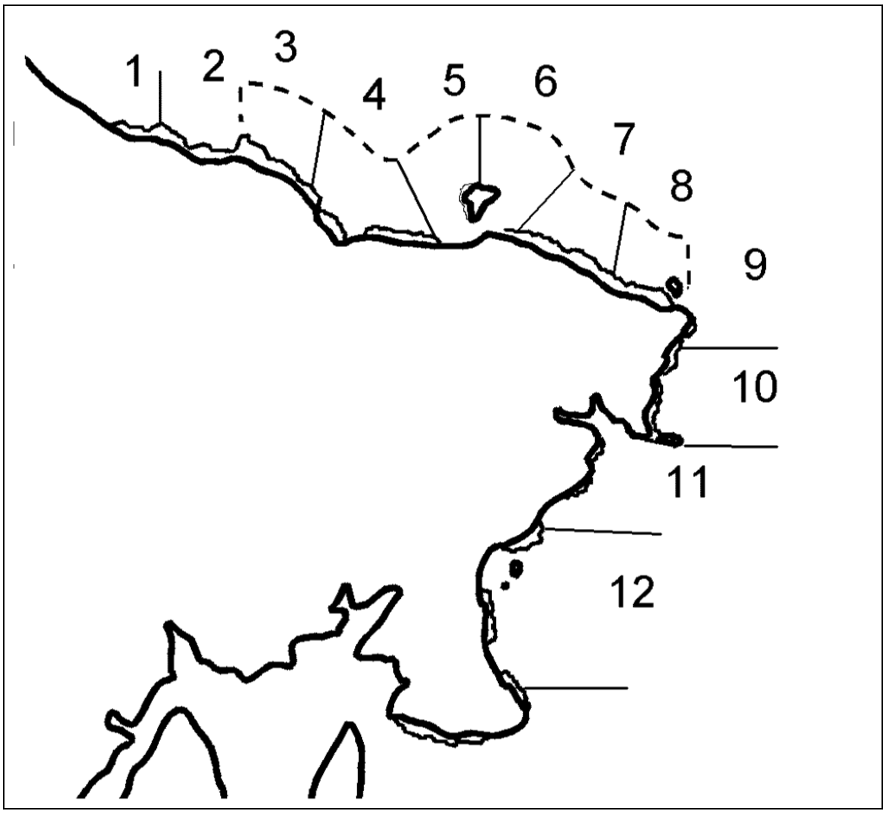

Figure 1. Map of sampling areas in and around the Cape Rodney to Okakari Point Marine Reserve. The dashed line shows the reserve boundary.

Survey methods

Underwater visual census

Within each site, two divers surveyed fishes within a total of ten 25 m × 5 m transects. A diver would fasten a fibreglass tape to the substratum, then swim 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. Occasionally, blue cod would follow divers between transects, and care was taken not to include these individuals in subsequent transect replicates. Depth and broad habitat type were recorded for each transect.

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 minutes duration (from the time the unit contacted the seabed).

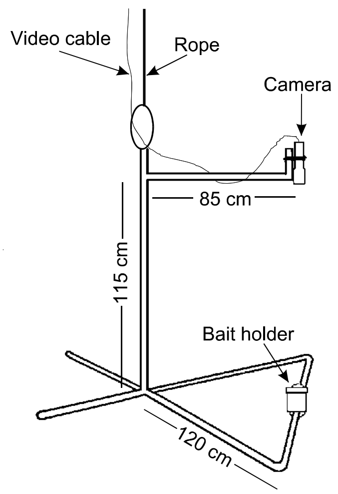

Figure 2. Baited underwater video assembly, with dimensions of the stand.

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 2000b, Willis et al. 2000). Individual fish were measured (fork length for snapper, total length for blue cod) 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., LEGsna, JUVsna).

Statistical analyses

Univariate analyses

Three univariate variables were of particular interest from the BUV data: the density of snapper (i) of all sizes, (ii) of legal size (> 270 mm fork length) and (iii) juveniles (< 270 mm fork length). These variables consist of ‘count’ data, which 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). Therefore, as in previous reports, ratios of densities of snapper between reserve and non-reserve areas for BUV data were analysed using generalised linear models (GLMs, McCullagh & Nelder 1989). Count data for each variable of interest was modelled using the log-linear model with overdispersed Poisson errors, 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 computer package (R Development Core Team 2004). This expresses the fish counts, *Y*, as

$$Y \sim Poisson(\lambda)$$

where *Poisson*(λ) denotes a (possibly overdispersed) Poisson distribution with expected value of λ, and log(λ) is modelled as a linear function of the factors. 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

$$\log(\lambda_{ij}) = \mu + \alpha_i$$

where μ is the overall mean and α is the parameter corresponding to the status effect to be estimated. For a log-linear model, the estimates of effects are multiplicative in nature. 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).

In addition to the multiplicative models obtained using the GLM approach as described above for the BUV data, there were also cases where specific contrasts were of interest. These were tested using non-parametric approaches. For example, we wished to contrast the UVC (or BUV) observations for individual species of fish (snapper and blue cod) for the previous census (autumn 2003) with those obtained in the current year (autumn 2005). This was done using the means of the observations from each area (because the error variability for the test was considered to be that from area to area) and performing a paired Wilcoxon signed rank test (e. g., Sokal and Rohlf 1981), with continuity correction. Such an approach is appropriate for non-normal data. Another specific contrast of interest for particular variables was that between the area means inside versus those outside the marine reserve for the UVC data. Here, the two-sample Wilcoxon rank sum test (equivalent to the Mann-Whitney U test, see Sokal and Rohlf 1981), again with continuity correction, was used.

The total number of species and the total number of individuals recorded using UVC were also analysed using a traditional two-way nested ANOVA, with “Status” (reserve versus non-reserve) treated as a fixed factor and “Areas” treated as a random factor, nested within “Status”. Levene’s test for homogeneity of variances and Shapiro-Wilk tests for normality ensured assumptions were fulfilled for each of these two variables before proceeding with the ANOVA. All non-parametric and traditional univariate tests were done using the R statistical computer program (R Development Core Team 2004).

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 27 fish species recorded and included in analyses and a total of 24 multivariate observations, consisting of 2 stations within each of 12 areas, with 6 areas located inside the reserve (areas 3-8) and 6 areas located outside the reserve (areas 1, 2, 9-12).

All multivariate methods were based on Bray-Curtis dissimilarities (Bray and Curtis 1957) calculated among observations for data transformed to $y' = \ln(y+1)$. Whole assemblages were analysed using permutational multivariate analysis of variance (PERMANOVA, Anderson 2001a), with "Status" (reserve versus non-reserve) treated as a fixed factor and "Areas" treated as a random factor, nested within "Status". P-values were obtained using appropriate permutation tests for each individual term in the model (Anderson 2001b). Data were also tested for homogeneity of multivariate dispersions using the computer programme PERMDISP (Anderson 2004). Relative dissimilarities in 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 and Willis 2003, Anderson and Robinson 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. Correlations of individual species with the canonical axis corresponding to "Status" was used as an indication of the species responsible for patterns of differences in assemblages observed between reserve and non-reserve stations. *P*-values for all multivariate tests (PERMANOVA, PERMDISP and CAP) were obtained using 9999 permutations.

Non-metric multidimensional scaling (nMDS) was used to display long-term changes at the community level. nMDS creates low-dimensional maps of relationships among samples (in this case each survey-status combination), where the distance between two points is proportional to their ranked biological dissimilarity as measured by a dissimilarity coefficient. The nMDS was done on density data for all taxa except the pelagic schooling species (yellow-eyed mullet *Aldrichetta forsteri*, kahawai *Arripis trutta*, koheru *Decapterus koheru*, and jack mackerel *Trachurus novaezelandiae*). Density data were averaged for each survey-status combination prior to analysis. The nMDS was run both with and without density data for the heavily harvested species (snapper and blue cod) in order to determine their influence on the overall community pattern.

RESULTS

Baited underwater video

Snapper Pagrus auratus

After four years in which total snapper densities within the reserve increased slowly from an average of ~12 individuals per BUV drop in autumn 1998 to 14.6 in autumn

2002, the autumn 2003 survey revealed a dramatic increase in mean density to 26.7 individuals per BUV drop. In autumn 2005 the mean density dropped from this high to 19.0 ± 3.8 (SE) fish per BUV drop (paired Wilcoxon signed rank test comparing autumn 2003 with autumn 2005: $P = 0.03$) (Fig. 3a, Table 1). Outside the reserve the mean total snapper density in 2005 was only 5.3 ± 1.4 individuals per BUV drop, a value that differed little from previous years (autumn densities ranged from ~3.5-6.7 during 1998-2003). Legal sized (> 270 mm fork length) snapper were 12.8 times more abundant inside the reserve than outside in 2005, down on the ratio of 27.7 observed in 2003, but in line with values from previous autumn surveys (10.4 in 2001 and 13.1 in 2002) (Fig. 3b, Table 1). Legal snapper continue to be rare outside the reserve (1.3 ± 0.4 individuals per BUV drop). Densities of undersize fish were consistent with previous years (2.5 ± 0.9 individuals per BUV drop inside the reserve and 4.0 ± 1.1 outside), and continued the trend of no consistent difference between reserve and non-reserve areas (Fig. 3c, Table 1).

The spatial distribution pattern of legal snapper was broadly consistent with earlier surveys in that the highest densities occurred near the centre of the reserve, around Goat Island (Fig. 4). Interestingly, the decline in densities of legal snapper from 2003 to 2005 appears to be due largely to a decrease in numbers in the western half of the reserve (areas 3-5). Densities in the reserve were very low at the western boundary (area 3) in 2005 but boundary area 8 at the eastern end of the reserve continued to hold reasonable numbers of fish. Snapper densities are naturally low in Area 1 because of the limited seaward extent of reef. Areas 2 and 9, however have considerable reef area, but are intensively fished both from boats and the shore (T. J. Willis, pers. obs.). High fishing pressure from these areas is likely to affect reserve areas 3 and 8.

As in previous surveys, the average fork length of snapper inside the reserve in 2005 was over 100 mm greater than that of fish outside the reserve (Fig. 5, Table 2). In 2005 we recorded the largest snapper ever seen on BUV in the reserve; this individual was at Alphabet Bay on the western side of Goat Island and had an estimated fork length of 1029 mm. A fish of this length would weigh 18.6 kg (or 41.1 lb) according to the length-weight equation of Paul (1976), though it may be dangerous to extrapolate to a fish this long as the largest of the 780 fish that contributed data to Paul's equation was only 710 mm FL.

Data from the four autumn surveys reveal a steady increase in the average size of all snapper within the marine reserve of 20-23 mm per year from 2001 to 2005 (Table 2). Fish outside the reserve showed comparable increases in average length from 2001 to 2003, but there was no further increase in 2005.

Table 1. Mean densities of snapper *Pagrus auratus* inside and outside the Cape Rodney to Okakari Point Marine Reserve, from 2000-2005 BUV surveys. Statistically significant (*P* < 0.05) ratios of reserve (R) to non-reserve (NR) densities are denoted by *. 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	9.00	7.57	1.19	0.62	2.28
	LEGsna	4.23	0.05	88.77*	4.78	1646.98
	JUVsna	4.77	7.52	0.63	0.30	1.35
Autumn 2001	MAXsna	13.42	6.67	2.01*	1.12	3.62
	LEGsna	7.79	0.75	10.39*	3.84	28.07
	JUVsna	5.62	5.91	0.95	0.47	1.91
Spring 2001	MAXsna	7.08	4.09	1.73	0.87	3.45
	LEGsna	6.17	0.87	7.09*	2.51	20.06
	JUVsna	0.91	3.22	0.28*	0.11	0.76
Autumn 2002	MAXsna	14.58	5.62	2.59*	1.49	4.52
	LEGsna	10.33	0.79	13.05*	4.47	38.10
	JUVsna	4.24	4.83	0.88	0.46	1.17
Autumn 2003	MAXsna	26.67	4.08	6.53*	4.12	10.36
	LEGsna	21.92	0.79	27.69*	11.56	66.32
	JUVsna	4.75	3.29	1.44	0.82	2.54
Autumn 2005	MAXsna	19.04	5.29	3.60*	2.17	5.96
	LEGsna	16.54	1.29	12.81*	5.89	27.85
	JUVsna	2.50	4.00	0.63	0.28	1.38

Table 2. Mean sizes of snapper *Pagrus auratus* inside and outside the Cape Rodney to Okakari Point Marine Reserve, from 2000-2005 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
All snapper						
Spring 2000	288.9	197	148.8	159	140.2*	24.9
Autumn 2001	307.7	322	203.5	160	104.1*	18.8
Spring 2001	389.2	165	217.9	94	171.3*	25.4
Autumn 2002	328.8	342	214.4	135	114.4*	19.1
Autumn 2003	351.6	640	242.1	98	109.5*	20.1
Autumn 2005	391.5	457	241.9	127	149.6*	22.7
Legal snapper						
Spring 2000	410.6	96	278.0	1	132.6	269.1
Autumn 2001	374.2	187	333.5	18	40.7	47.8
Spring 2001	410.5	145	310.0	21	100.4*	45.9
Autumn 2002	371.3	242	300.3	19	71.1*	45.5
Autumn 2003	377.4	526	343.2	19	34.2	40.1
Autumn 2005	417.8	397	294.6	31	123.2*	41.2

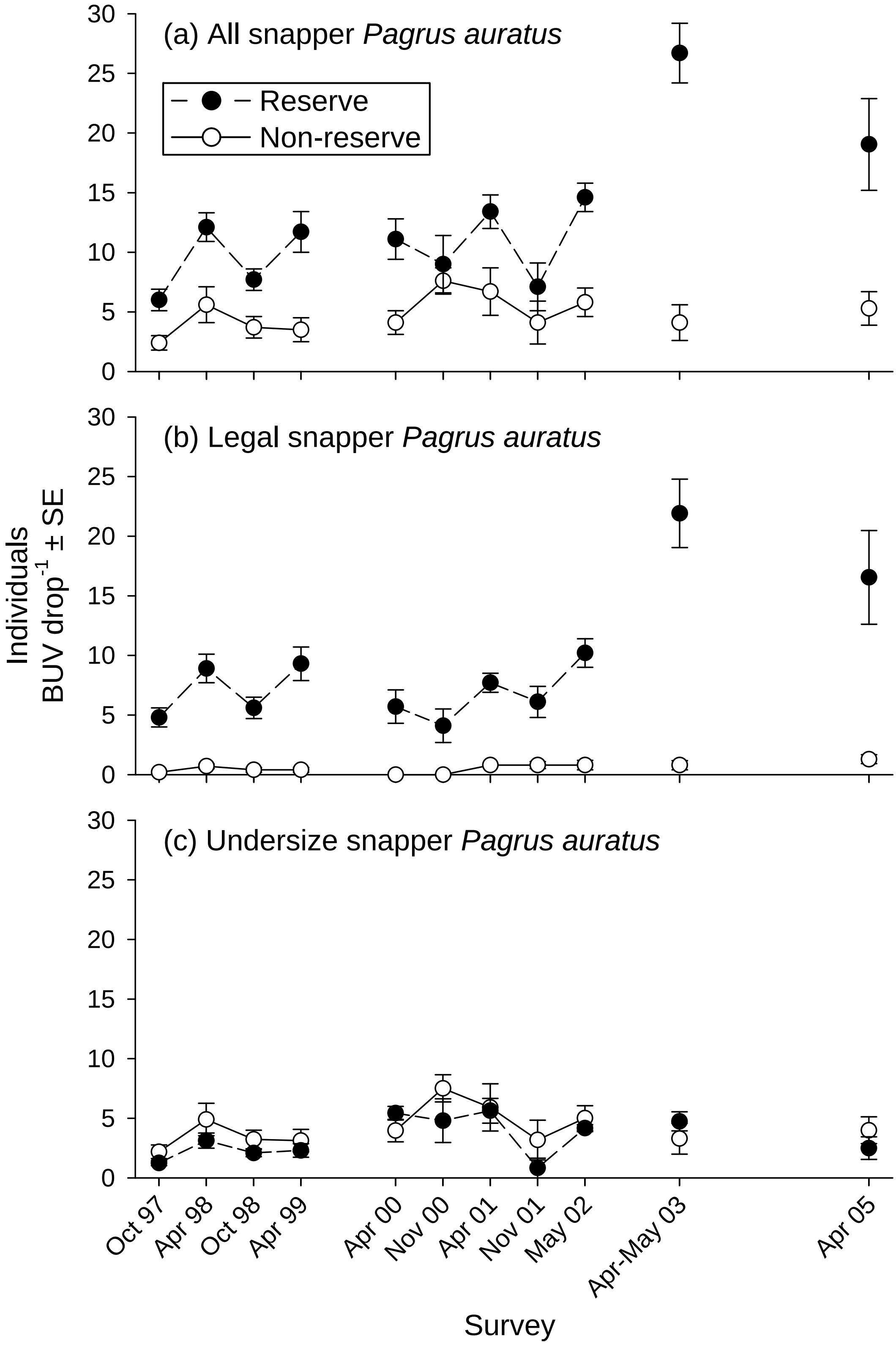

Figure 3. Long term trends in the relative density of snapper *Pagrus auratus* inside and outside the Cape Rodney to Okakari Point Marine Reserve, as measured using BUV. (a) All snapper (MAXsna), (b) Legal-size (> 270 mm fork length) snapper, (c) undersize snapper (< 270 mm fork length).

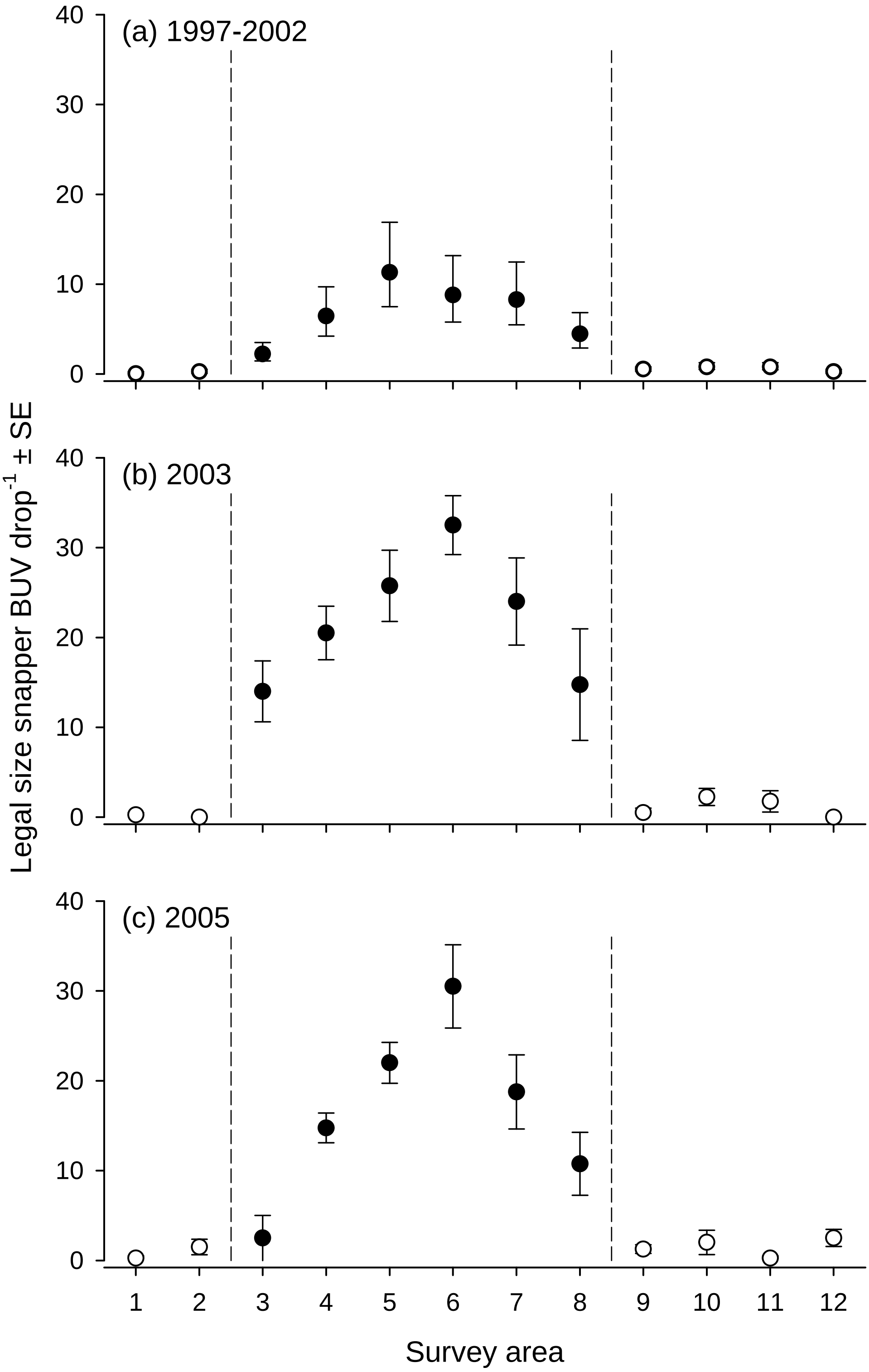

Figure 4. Relative density of legal-size snapper *Pagrus auratus* within the twelve survey areas, based on (a) modelled data from nine BUV surveys (October 1997–May 2002), and BUV data from 2003 (b) and 2005 (c). Closed symbols are within the reserve, open symbols are fished areas. Dashed vertical lines indicate reserve boundaries.

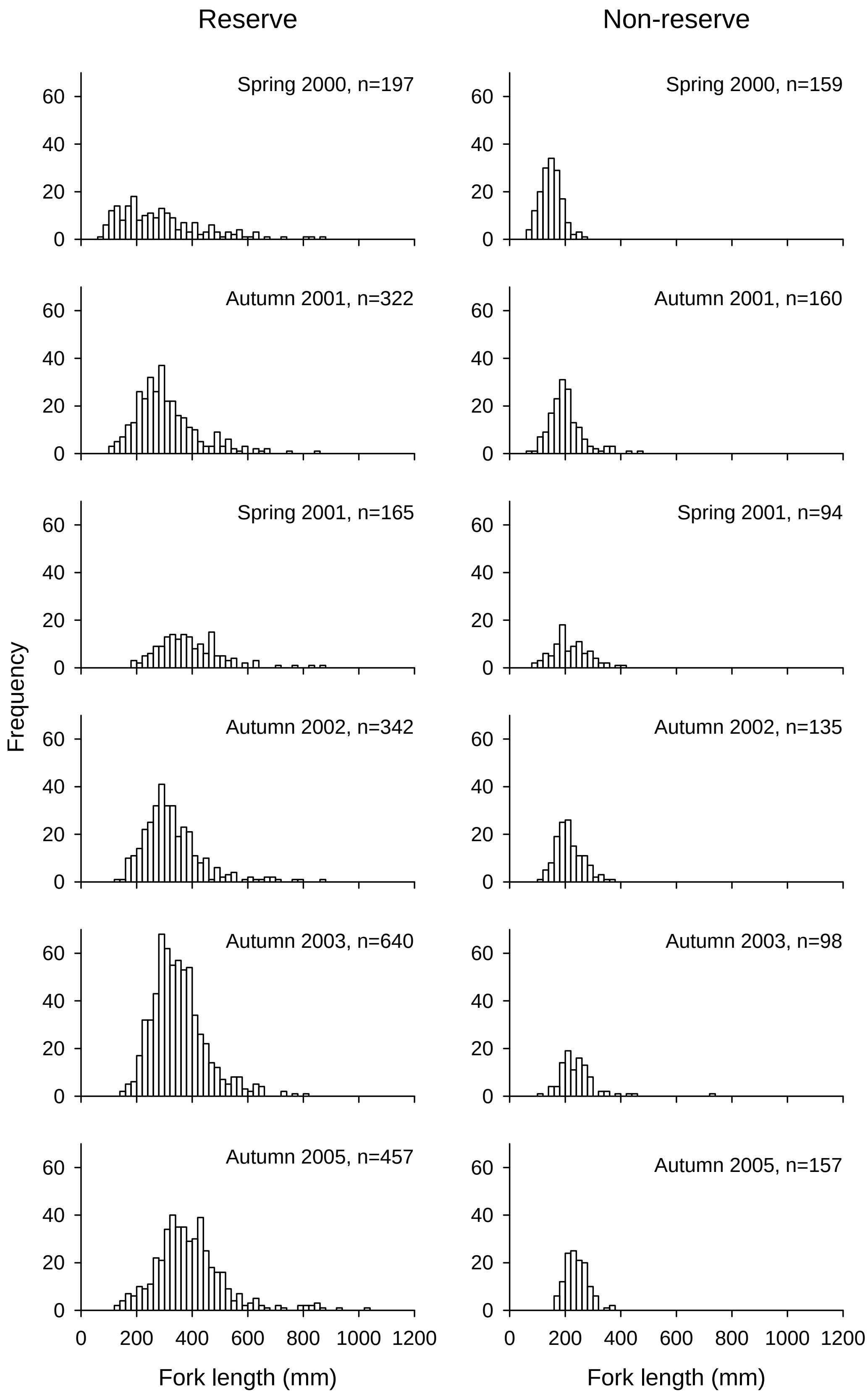

Figure 5. Size frequency distributions of snapper *Pagrus auratus* inside and outside the Cape Rodney to Okakari Point Marine Reserve from 2000-2005, as measured using BUV.

Blue cod Parapercis colias

As in previous years, the average blue cod density measured using BUV was higher inside the marine reserve than outside in 2005 (Fig. 6a, Table 3). The average density within the reserve in 2005 was similar to that in 2003, with no apparent trend for a return to the high measured in 1997 (Fig. 6a).

Willis et al. (2003b) suggested that the steep decline in average blue cod density from 1997 to 1999 might be attributable to increasing sea surface temperatures during that period (Fig. 6b). A much longer time-series is required to test the hypothesis that there is a negative correlation of blue cod densities with sea surface temperature, but the data thus far are consistent with it, in that densities declined during the warming period of 1997-1999, were constant while temperature was constant from 2000-2002, and have increased slightly while the temperature dropped from 2003-2005.

As previously, blue cod within the reserve were larger on average than those that occurred outside (Table 4), but numbers were too low for meaningful statistical analysis.

Table 3. Mean densities of blue cod *Parapercis colias* inside and outside the Cape Rodney to Okakari Point Marine Reserve, from 2000-2005 BUV surveys. Statistically significant ($P < 0.05$) ratios of reserve (R) to non-reserve (NR) densities are denoted by *.

Survey	Reserve mean	Non-reserve mean	R:NR ratio	Lower 95% CL for ratio	Upper 95% CL for ratio
Spring 2000	0.64	0.14	4.45*	0.94	21.08
Autumn 2001	0.50	0.04	12.00*	2.02	71.36
Spring 2001	0.46	0.00	∞*		
Autumn 2002	0.42	0.13	3.33*	1.22	9.90
Autumn 2003	0.79	0.00	∞*		
Autumn 2005	0.88	0.17	5.25*	1.42	19.40

Table 4. Mean sizes of blue cod *Parapercis colias* inside and outside the Cape Rodney to Okakari Point Marine Reserve, from 2000-2005 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
Spring 2000	314.0	14	242.7	4	71.2	75.8
Autumn 2001	257.2	12	117.0	1	140.2	-
Spring 2001	282.9	11	-	0	-	-
Autumn 2002	257.6	11	197.7	3	60.0	66.6
Autumn 2003	322.9	19	-	0	-	-
Autumn 2005	284.2	21	259.8	4	24.5	90.33

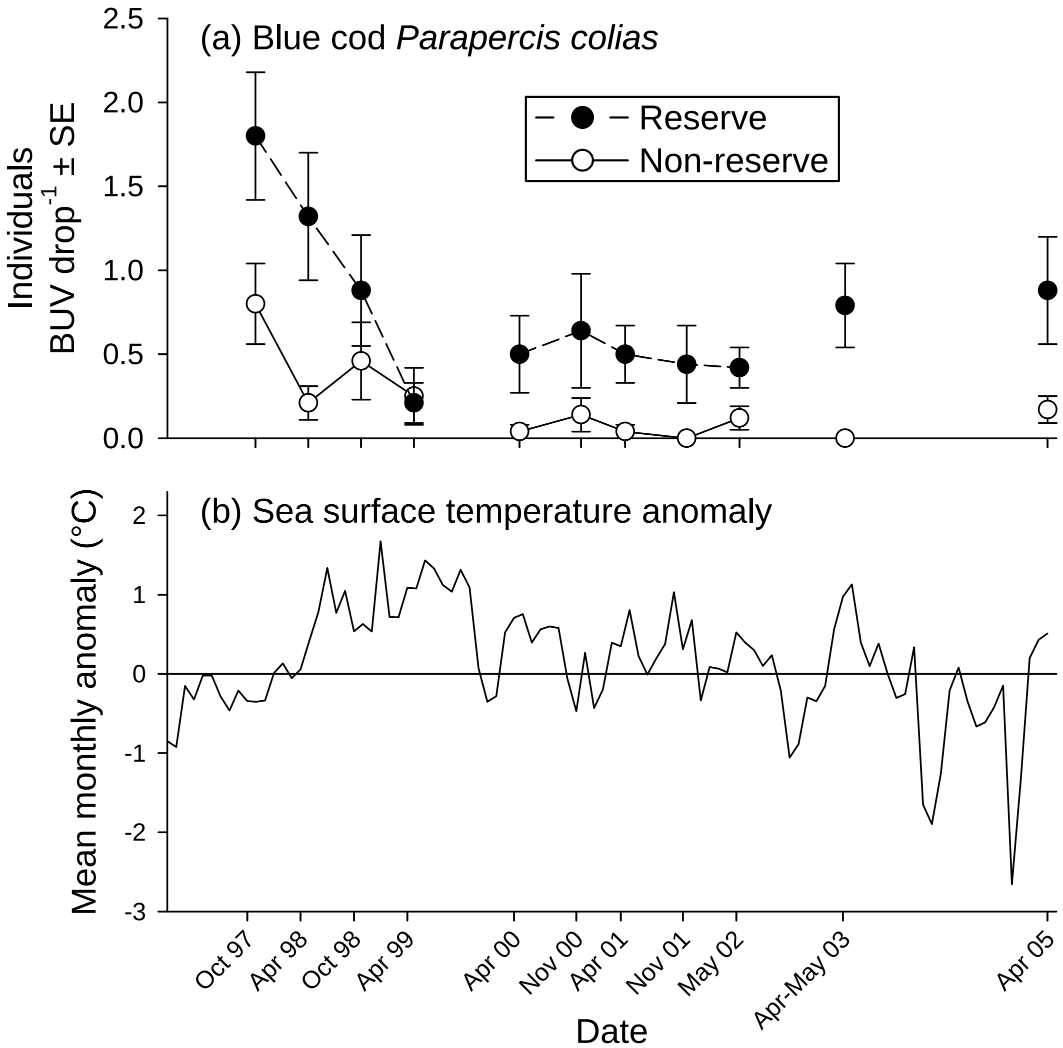

Figure 6. (a) Long term trends in the density of blue cod *Parapercis colias* inside and outside the Cape Rodney to Okakari Point Marine Reserve, as measured using BUV. (b) Sea surface temperature anomalies (from long term average 1967-96).

Underwater visual census

Community-level patterns

There was significant variability in fish assemblages observed by UVC from one area to the next in the study (Table 5, $P = 0.0045$). Over and above this variation, there was also some weak evidence for an effect of reserve status on fish assemblages (Table 5, $P = 0.0565$). However, this was apparently due largely to a difference in dispersion, with assemblages outside the reserve being more variable than those inside the reserve (Table 6, Fig. 7a). The mean Bray-Curtis dissimilarity among assemblages in non-reserve areas was 37%, compared to a mean dissimilarity of only 26% among assemblages inside the reserve. This might simply be due to the larger geographic spread of the non-reserve areas, which may therefore encompass a greater diversity of habitat as a consequence (e. g., fish assemblages in area 1, lying to the north of the reserve, are quite dissimilar to those occurring in area 11, lying to the south of the reserve, Fig. 7b).

Greater variability in fish assemblages outside the reserve might also be due to patchiness in the occurrence of species. There were, on average, significantly fewer species observed in areas outside the reserve (mean = 10.6 ± 0.50 SE), compared to areas inside the reserve (mean = 12.42 ± 0.50 SE) ($F_{1,10} = 6.37$, $P = 0.027$). This difference in diversity was apparent, even though the total abundance of fish observed by UVC (excluding pelagic species) did not vary significantly among areas ($F_{10,12} = 1.33$, $P = 0.317$). Although the mean total abundance of fish recorded by UVC was greater inside (mean = 316.33 ± 79.94 SE) compared to outside the reserve (mean = 262.67 ± 36.13 SE), this difference was not statistically significant ($F_{1,10} = 0.43$, $P = 0.524$).

The canonical analysis detected a clear and significant overall effect of reserve status on the fish assemblages (Fig. 8, canonical correlation, $\delta^2 = 0.750$, $P = 0.0055$). Leave-one-out allocation of observations on the basis of the CAP model resulted in an 83.33% success rate: twenty out of the twenty-four UVC pooled observation units were correctly classified as belonging to either the non-reserve or reserve type of assemblage. Several species were indicative of fish assemblages inside reserves and were more abundant there, including snapper, silver drummer, butterfish, blue cod, porae and sweep (Table 7). In contrast, assemblages outside reserves were indicated by greater numbers or frequencies of occurrence of spotty, hiwihiwi, trevally, marblefish, demoiselles, eagle rays and goatfish.

Long-term changes at the community level are shown in a non-metric multidimensional scaling (MDS) plot (Fig. 9a). The stress value of 0.10 indicates that the MDS mapped the points satisfactorily in two-dimensional space (values of stress < 0.2 are usually considered acceptable). In this plot the communities from inside and outside the reserve have clearly been distinct since autumn 2000, despite relatively large changes in community composition in the autumn 2002 and autumn 2003 surveys. As indicated in the other multivariate analyses, snapper and blue cod were not solely responsible for the reserve effect, since the communities remained fairly distinct when these two species were excluded from the analysis (Fig. 9b).

Table 5. PERMANOVA on the basis of the Bray-Curtis dissimilarity for ln(y+1) transformed species abundance data (27 species). P-values were obtained using permutation of appropriate units (Anderson 2001b).

Source	df	SS	MS	*F*	*P*
Status	1	2017.85	2017.85	1.88	0.0565
Areas(Status)	10	10730.17	1073.02	1.82	0.0045
Residual	12	7058.22			
Total	23	19806.24			

Table 6. PERMDISP on the basis of the Bray-Curtis dissimilarity for ln(y+1) transformed species abundance data (27 species). Note that when there are two observations per group, they will be an equal distance from their group centroid. Thus, there is no measured variance in the within-group dispersions when there are only two levels per group. This is why the SS is equal to zero for the residual below, as there were only two stations per area. This does not, however, preclude the analysis of differences in average dispersion between reserve and non-reserve areas, as shown below.

Source	df	SS	MS	*F*	*P*
Status	1	153.07	153.07	6.71	0.0291
Areas(Status)	10	228.27	22.83	---	---
Residual	12	0.00	0.00		
Total	23	381.34			

Table 7. Individual species having correlations of $|r| > 0.20$ with the canonical axis separating reserve from non-reserve sites and occurring in at least 10% of the sites.

Positive correlation (reserve)		*r*
Snapper	*Pagrus auratus*	0.677
Silver drummer	*Kyphosus sydneyanus*	0.571
Butterfish	*Odax pullus*	0.446
Blue cod	*Parapercis colias*	0.385
Porae	*Nemadactylus douglasii*	0.374
Sweep	*Scorpis lineolatus*	0.261
Negative correlation (non-reserve)		
Spotty	*Notolabrus celidotus*	-0.639
Hiwihiwi	*Chironemus marmoratus*	-0.405
Trevally	*Pseudocaranx dentex*	-0.388
Marblefish	*Aplodactylus arctidens*	-0.290
Demoiselles	*Chromis dispilus*	-0.290
Eagle Ray	*Myliobatus tenuicaudatus*	-0.222
Goatfish	*Upeneichthys lineatus*	-0.208

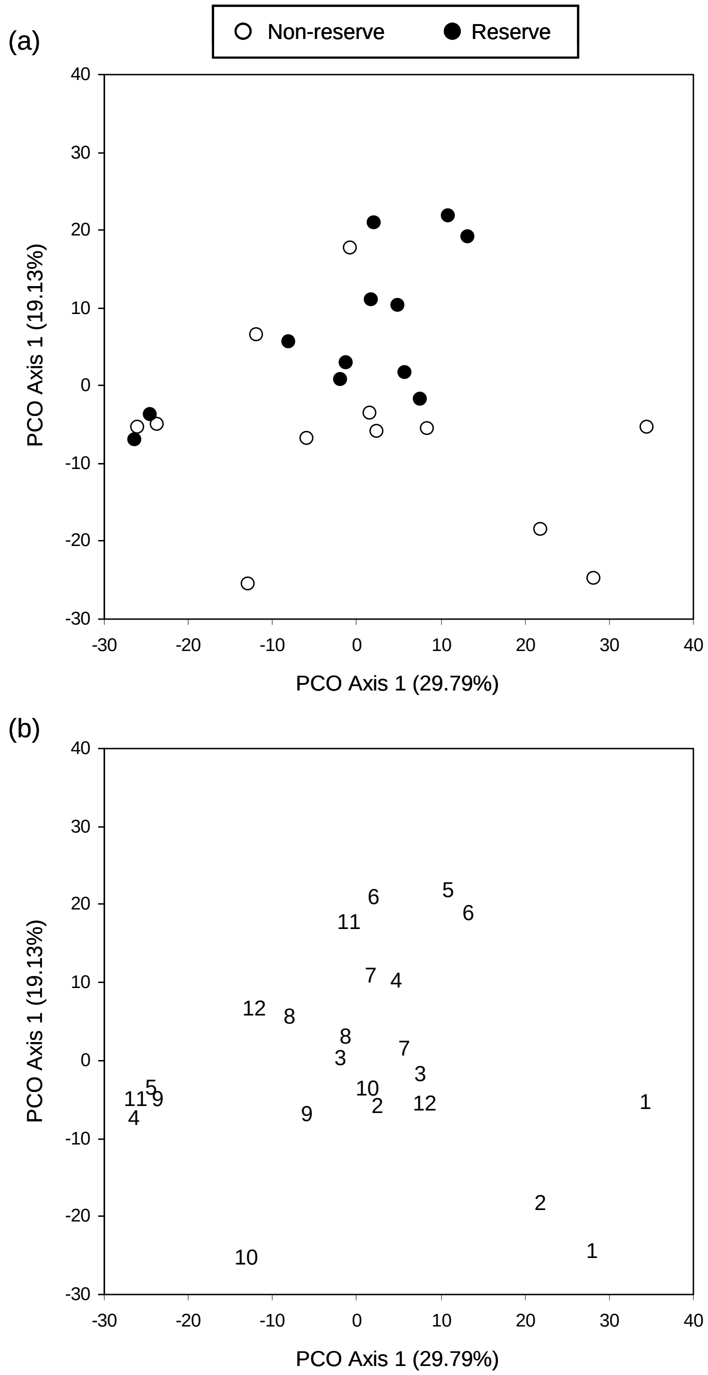

Figure 7. Ordination plot of the first two PCO axes (explaining 48.92% of the original variability) based on Bray-Curtis dissimilarities of ln(y+1) transformed species abundance data (27 species), showing assemblages at different stations with labels for (a) reserve versus non-reserve status or (b) areas 1 through 12 (with 2 stations per area).

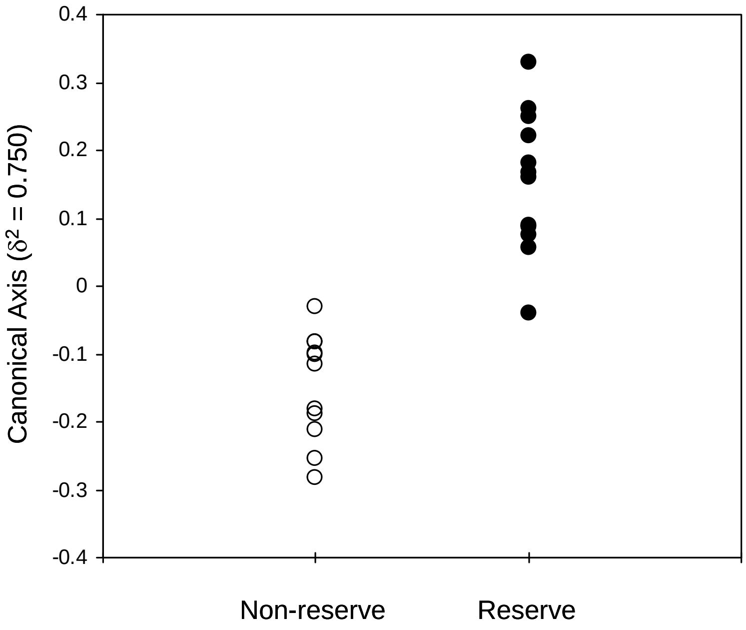

Figure 8. Plot of the canonical axis from a CAP constrained ordination to discriminate fish assemblages from reserve versus non-reserve stations. The discriminant analysis was done on the first m = 9 PCO axes (which explained 99.99% of the original variability) from Bray-Curtis dissimilarities of ln(y+1) transformed species abundances (27 species).

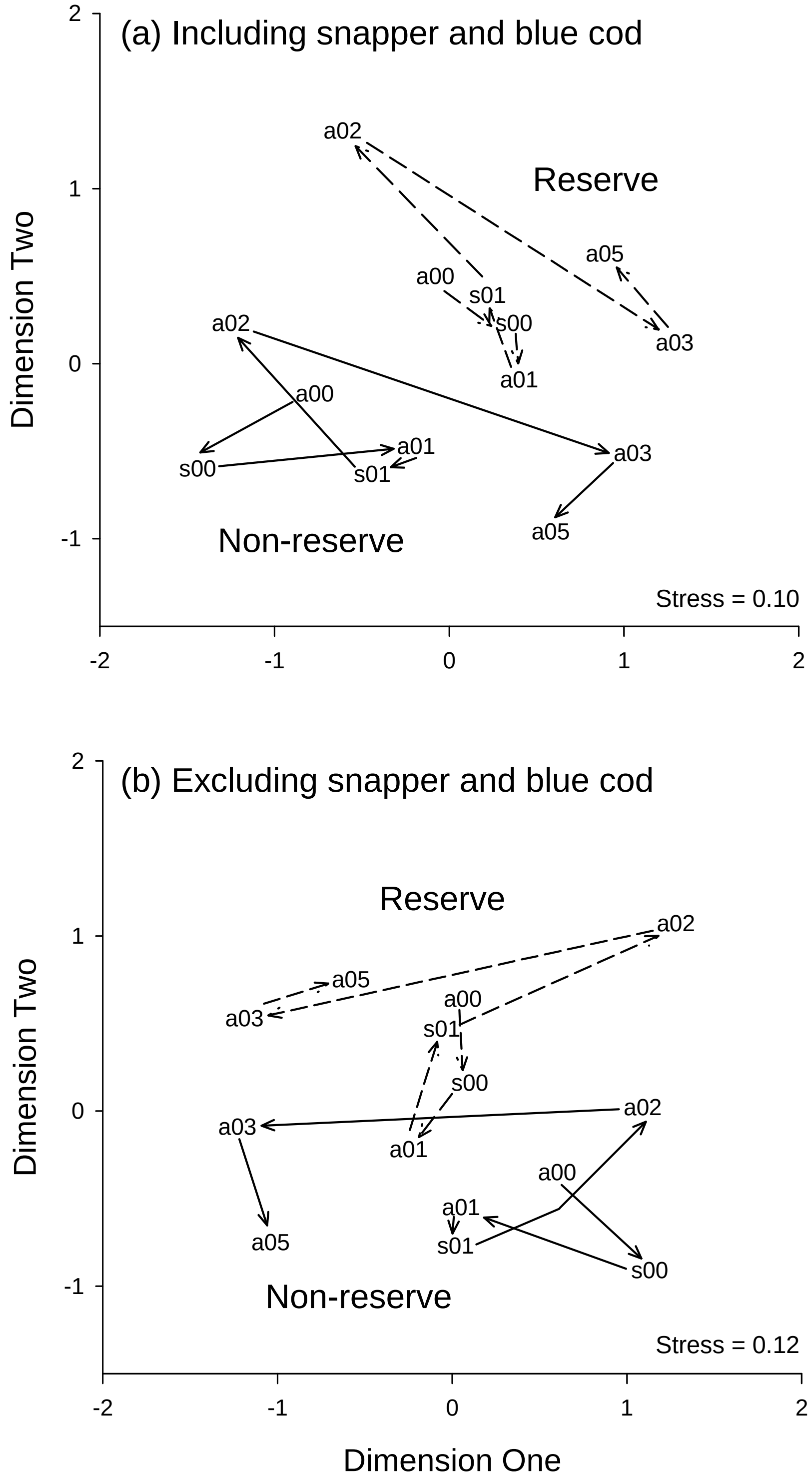

Figure 9. Non-metric multidimensional scaling (MDS) plot of changes in the fish communities inside and outside the Cape Rodney to Okakari Point Marine Reserve, with (a) and without (b) heavily harvested species. Pelagic species were excluded from both analyses. Codes correspond to season and year, e. g., a00 = autumn 2000, s00 = spring 2000. The MDS was constructed from a Bray-Curtis dissimilarity matrix based on log (y+1) transformed abundance data.

Individual species

In autumn 2005 the UVC estimate of the mean density of snapper within the reserve was slightly lower than the 2003 value (3.09 vs 3.97 individuals per 125 m^2; paired Wilcoxon signed rank test: P = 0.56) (Fig. 10), reflecting the pattern seen in the recent BUV data (Fig. 3a). Snapper densities continue to be higher inside the reserve than outside, by a factor of 4.1 times in 2005 (3.09 vs 0.75 individuals per 125 m^2; two-sample Wilcoxon rank sum test: P = 0.031). Blue cod displayed similar patterns to snapper, with mean densities in the reserve decreasing slightly from 2003 to 2005 (from 0.46 to 0.25 individuals per 125 m^2), although this increase was not detected as statistically significant (paired Wilcoxon signed rank test: P = 0.06, Fig. 10). In 2005 blue cod densities were a non-significant 2.0-fold higher inside the reserve than outside (0.25 vs 0.13 individuals per 125 m^2) (two-sample Wilcoxon rank sum test: P = 0.31). Red moki densities continued to be similar inside and outside the reserve.

The relative densities of spotty and banded wrasse followed very similar patterns in time (Fig. 11). Densities were generally higher outside the reserve. The relative density of trevally was variable, as expected for a schooling species (Fig. 11). As in 2003, very few trevally were recorded in 2005, but this is probably a function of their patchiness – although no individuals were recorded in autumn 2003 at least a hundred individuals were seen near Shag Rock at Goat Island in July 2003 (Taylor et al. 2003).

Average densities of leatherjackets and goatfish continued to show no consistent differences between reserve and non-reserve areas (Fig. 12). Silver drummer continued to be more abundant inside the reserve (Fig. 12).

Average densities of sweep, blue maomao, and demoiselle, all schooling planktivores, varied less than might be expected for schooling species, but none showed consistent differences with respect to reserve status (Fig. 13).

Not surprisingly, densities of kahawai, jack mackerel and parore, all highly mobile schooling species, did not differ in a consistent manner between reserve and non-reserve areas (Fig. 14).

UVC estimates of butterfish mean densities within the reserve have fluctuated greatly from year to year, with no clear reserve effect (Fig. 15). This species responds negatively to divers, so its abundance was probably underestimated.

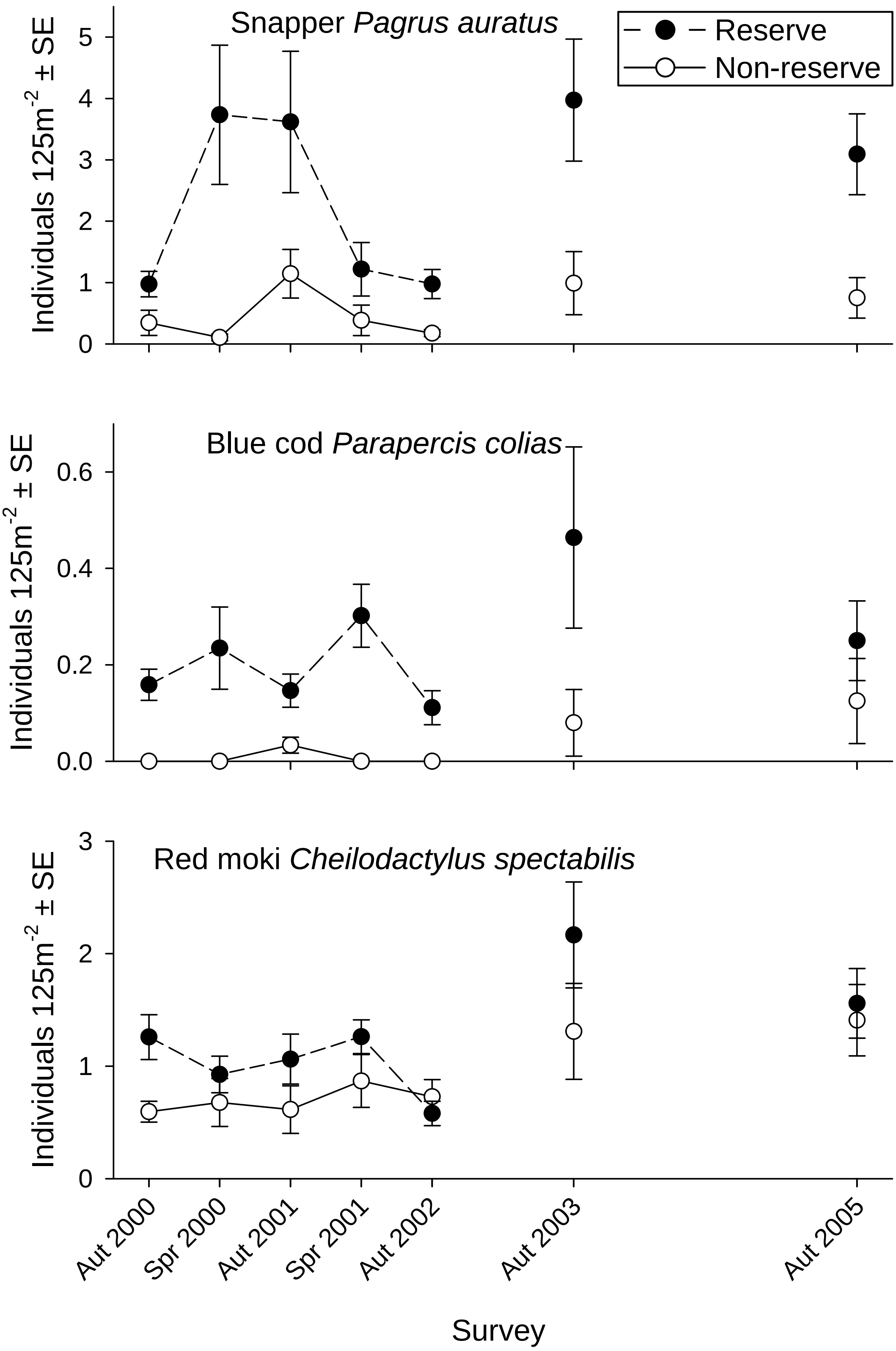

Figure 10. Long term trends in the densities of snapper, blue cod, and red moki inside and outside the Cape Rodney to Okakari Point Marine Reserve, as measured using UVC.

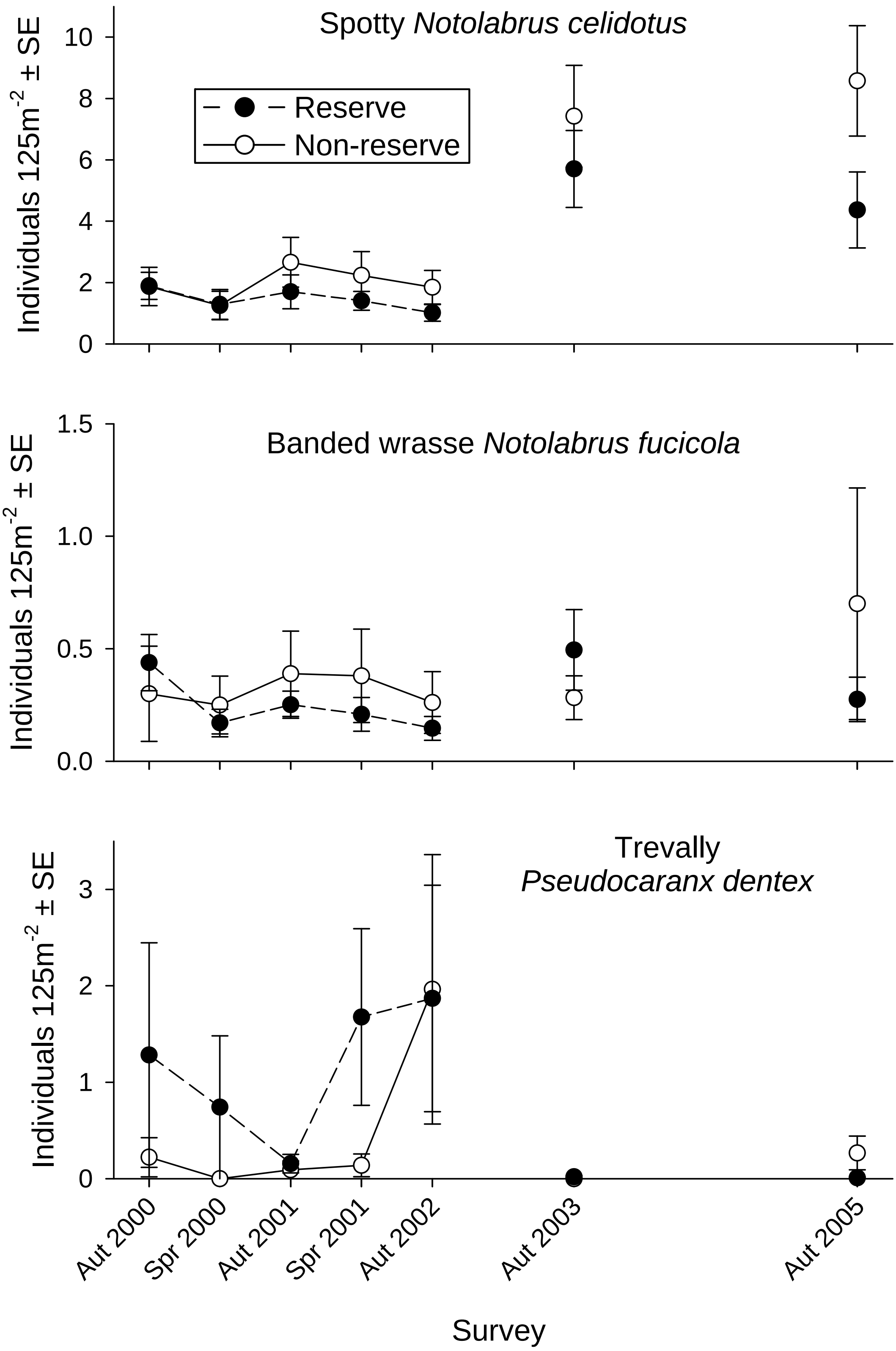

Figure 11. Long term trends in the densities of spotty, banded wrasse, and trevally inside and outside the Cape Rodney to Okakari Point Marine Reserve, as measured using UVC.

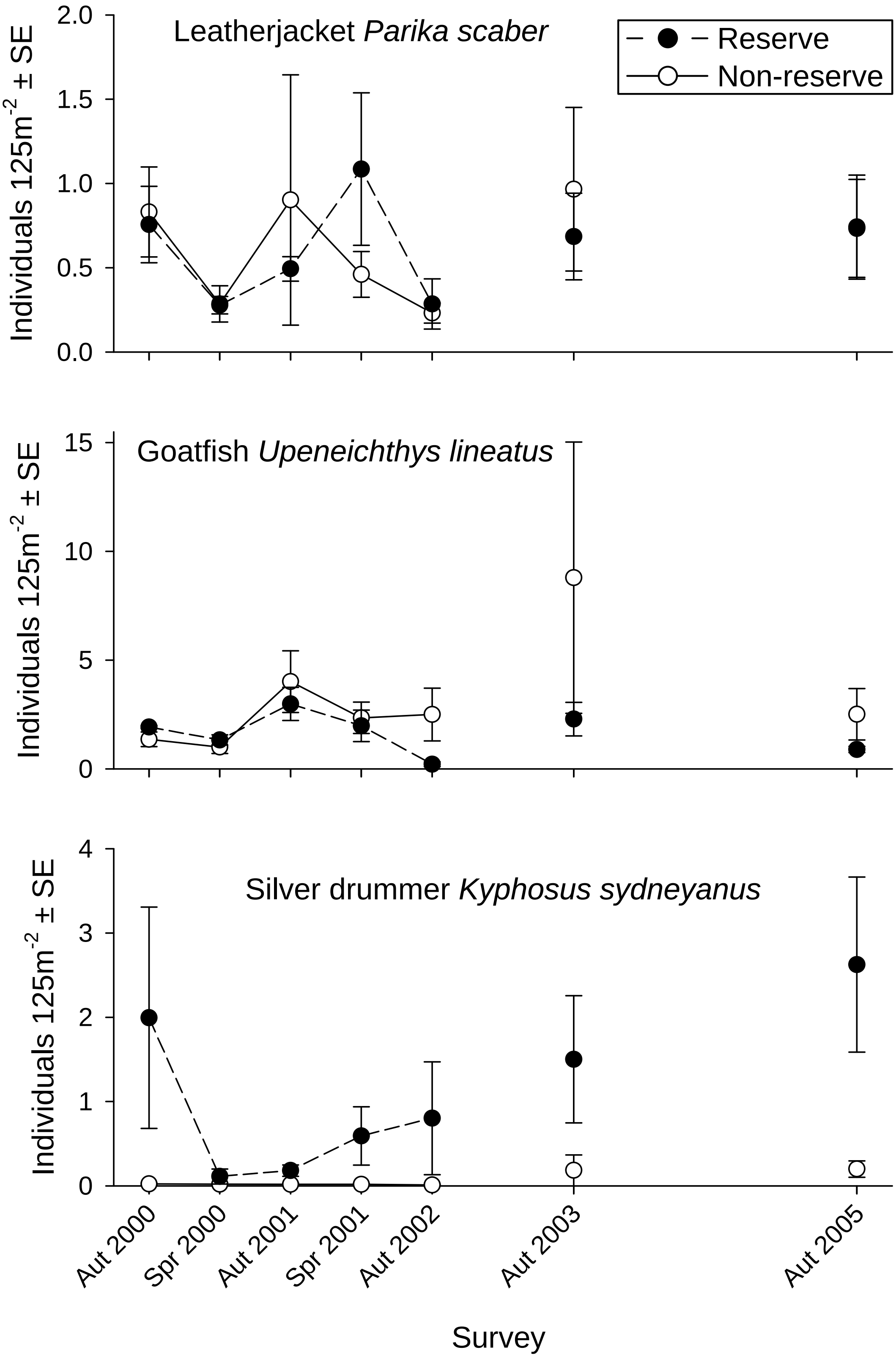

Figure 12. Long term trends in the densities of leatherjacket, goatfish, and silver drummer inside and outside the Cape Rodney to Okakari Point Marine Reserve, as measured using UVC.

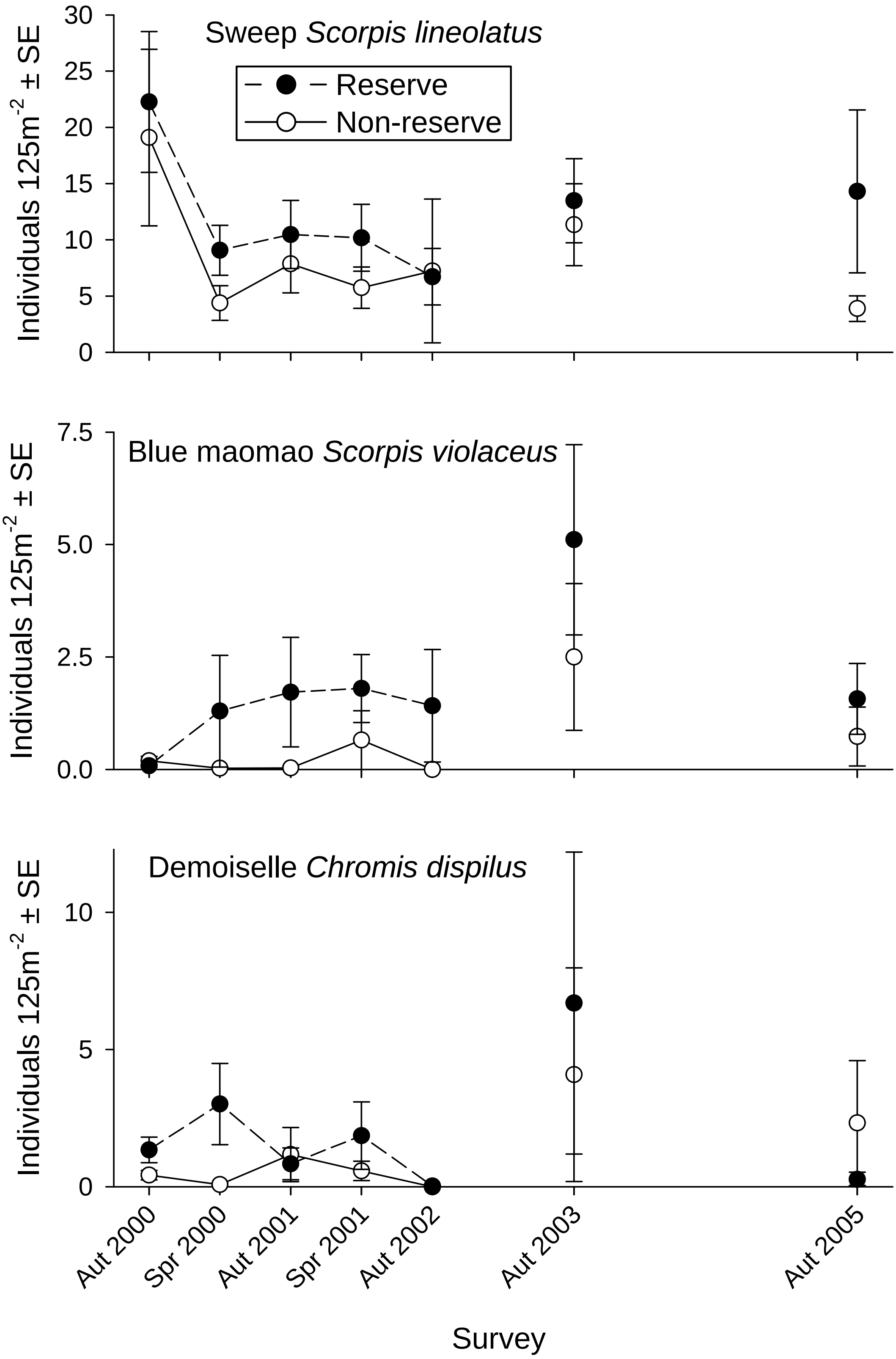

Figure 13. Long term trends in the densities of sweep, blue maomao, and demoiselle inside and outside the Cape Rodney to Okakari Point Marine Reserve, as measured using UVC.

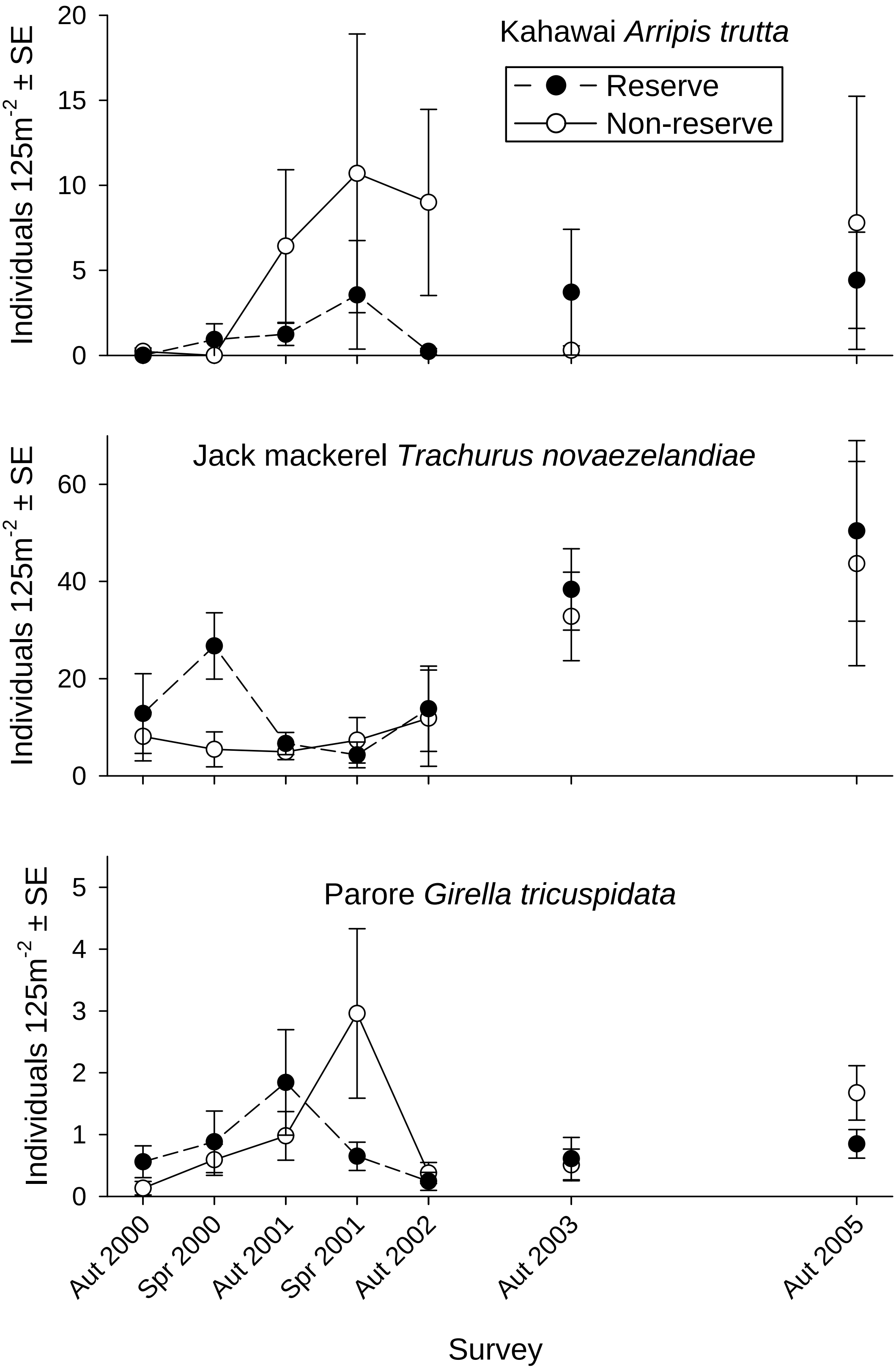

Figure 14. Long term trends in the densities of kahawai, jack mackerel, and parore inside and outside the Cape Rodney to Okakari Point Marine Reserve, as measured using UVC.

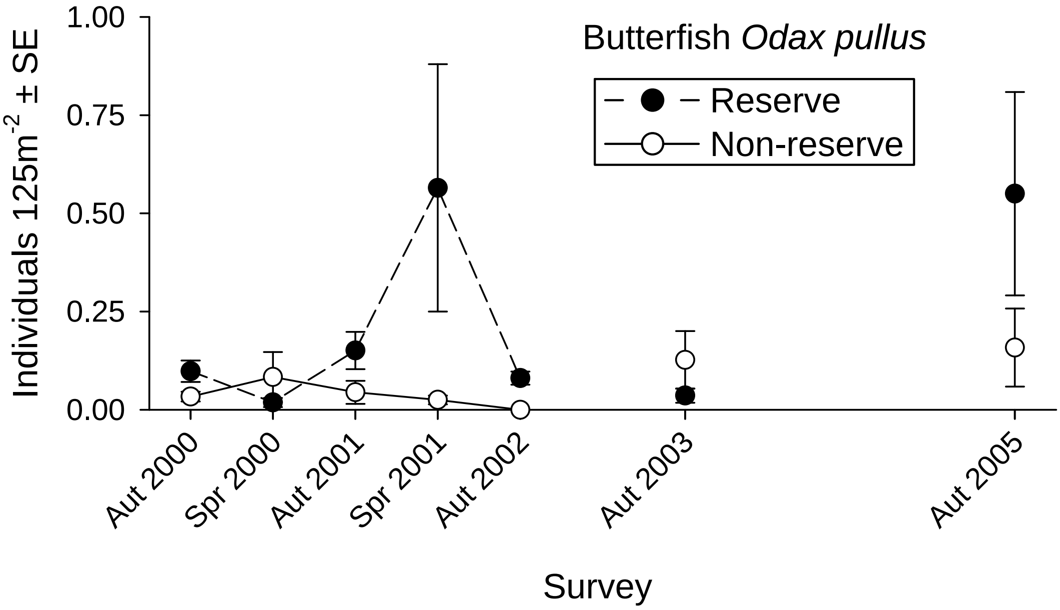

Figure 15. Long term trends in the densities of butterfish inside and outside the Cape Rodney to Okakari Point Marine Reserve, as measured using UVC.

DISCUSSION

As in past surveys, the autumn 2005 survey upon which this report is based revealed differences between fish communities living in the Cape Rodney to Okakari Point Marine Reserve and those in adjacent fished areas. These differences have been, and continue to be, particularly evident for those fish species experiencing heavy fishing pressure outside the reserve: the snapper *Pagrus auratus* and the blue cod *Parapercis colias*.

The autumn 2003 BUV survey detected an approximate doubling in numbers of legal-sized (> 270 mm fork length) snapper within the reserve since the previous survey in autumn 2002, which was attributed to an exceptionally large influx of individuals from offshore waters (Taylor et al. 2003). The increase from 2002 to 2003 followed a 5 year period in which average autumn densities were relatively stable, though there was a suggestion of a small but steady increase from 2000 to 2002. The autumn 2005 BUV survey found that densities had dropped from the 2003 high to a level in line with the the aforementioned upward trend from 2000 to 2002. Snapper densities within the reserve clearly have not yet stabilised, and indeed may never be stable given the apparent importance of seasonal movement in and out of the reserve, and the consequential influence of variation in the wider stock due to factors such as recruitment and fishing.

The spatial distribution pattern of legal snapper was broadly consistent with earlier surveys in that the highest densities occurred near the centre of the reserve, but the drop-off in numbers in the western part of the reserve in 2005 was much steeper than in 2003 (this accounted for a large part of the decline in total densities since 2003).

There is no obvious explanation for this pattern. As noted in previous reports, the lack of any apparent plateau in densities around the centre of the reserve possibly indicates that the majority of the reserve is being affected by the loss of snapper to fishing at the boundaries, although alternative explanations are that the central region of the reserve contains a more favourable environment for snapper in terms of reef extent and topography, and/or more food provided by the public.

As for previous years, in 2005 the average blue cod density measured using BUV was higher inside the marine reserve than outside. The average density within the reserve in 2005 was similar to that in 2003, with no apparent trend for a return to the high measured in 1997. Willis et al. (2003b) suggested that the steep decline in average blue cod density from 1997 to 1999 might be attributable to increasing sea surface temperatures during that period, because blue cod are essentially a “southern” species presumably better suited to cooler waters. A much longer time-series is required to test the hypothesis that there is a negative correlation of blue cod densities with sea surface temperature, but the data thus far are broadly consistent with it, in that densities declined during the warming period of 1997-1999, were constant while temperature was constant from 2000-2002, and have increased slightly while the temperature dropped from 2003-2005.

It is extremely difficult to predict the future trajectory of the fish community within the reserve, due to the large number of potential influences, their unpredictability, and the complexity of interactions among these factors. For example, the intensity of fishing outside the reserve undoubtedly has a major effect on the abundance of species like snapper and blue cod within the reserve (evident in the boundary effects), and may increase in future years as Auckland moves northward and fishing pressure increases. Long-term climatic changes have the potential to affect fishes in a variety of ways. One of the most important will be the potential effects on current patterns and upwelling, which may affect the transport of larvae to reefs, the productivity of planktonic larval food sources, and thus juvenile recruitment (Cushing 1995). Habitat changes due to cascading effects of increased densities of predators such as snapper have potentially important consequences for a range of fish species, and have been tentatively implicated in the recent decline of blue cod (Willis et al. 2003b). In this light it is interesting to note that species having consistently positive correlations with non-reserve areas (*Notolabrus celidotus* and *Chironemus marmoratus*; Table 7, also see equivalent analyses for previous years in Willis et al. 2003b, Taylor et al. 2003) are typically associated with the urchin barrens habitat (Anderson & Millar 2004), which is now more widespread outside the reserve due to intense predation on sea urchins inside the reserve (Shears & Babcock 2002). However, higher densities of *Kyphosus sydneyanus* inside the reserve are difficult to explain given that this species also is typically associated with urchin barrens habitat (Anderson & Millar 2004). *Kyphosus sydneyanus* is widely considered inedible by humans, so is unlikely to be targeted outside the reserve, though it is possibly taken in gill nets set for lobster bait.

General UVC surveys are useful for making broad comparisons and detecting large changes in fish assemblages, and can determine differences between reserves and fished areas, but can generate as many questions as they provide answers. Different species occupy different habitats, have different modes of behaviour (e. g., solitary versus schooling), and respond to divers in different ways. If it is a priority for DoC to determine whether marine reserves can mitigate the effects of fishing on species such as blue maomao, trevally or butterfish, then surveys must be done using methods tailored to those species, much as specific methods (BUV) were needed to assess the relative density of snapper. Habitat will need to be taken into account for some species (e. g., butterfish), while survey techniques may need to be modified in other ways for others (e. g., pelagic or demersal schooling species). The absence of trevally and butterfish from the list of species showing a strong positive effect of reserve protection in 2003 is probably due to the difficulty of sampling these patchily distributed species, rather than to a lack of response.

Optimisation of techniques for surveying targeted species should be undertaken as part of a separate programme that would have nationwide benefits, and will pay major dividends if addressing well-defined conservation needs. For example, development of methods to survey schooling species has important applications at high-profile diversity hotspots such as the Poor Knights Islands or Tuhua.

The recent variation in snapper densities underlines the need for continued regular monitoring of fish in the marine reserve. Snapper predation has major effects on rocky reef habitat structure via a trophic cascade involving sea urchins and seaweeds (Babcock et al. 1999, Shears & Babcock 2002, 2003), so higher snapper densities have potentially far-reaching impacts. Densities of red moki were once more than twice as high inside the reserve as outside (McCormick & Choat 1987), and now are not. Blue cod and spiny lobster (Kelly & Haggitt 2002) have declined since 1997 and 1995, respectively. Reserves are not static entities, and ongoing monitoring is really required in order to maintain up-to-date knowledge of trends, stability, potential impacts and measures of natural variation in these systems over longer periods of time.

Recommendations

- The fish monitoring programme should be continued at one to two year intervals with the current levels of sample replication regarded as a minimum level of effort.
- The programme should be extended to include comparison with projected new reserves (e. g., Great Barrier Island) using identical sampling design and methodology. Comparison of this established reserve with a new reserve will help elucidate the effects of protection on species that are not targeted by fishers.
- Such studies can only be achieved with a long-term commitment to monitoring. Any attempt to monitor new reserves should begin at least two years prior to reserve implementation and continue for at least five years afterward. The programme can then be reviewed based on (1) any changes observed, (2) the rate of such changes, and (3) the degree of seasonal and annual variability observed.
- The increasing number of surveys likely to be needed in an expanded network of marine reserves in New Zealand will require a more consistent and long-term approach to funding monitoring at regional and national scales, as well as the methodology and personnel to conduct it. Inconsistencies in methods and approach at different reserves would make the results difficult, if not impossible, to compare. Failure to address these issues will compromise the effectiveness of marine reserve monitoring nationwide.

ACKNOWLEDGEMENTS

Field work was completed with the assistance of Hawere skipper Brady Doak and divers Charles Bedford, Pam Brown, and Nick Shears. Thanks to Jo Evans for continuing to record and compile Leigh climate data.

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