CAPE RODNEY TO OKAKARI POINT MARINE RESERVE FISH MONITORING 2003: FINAL REPORT

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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 2003 and continues a time-series that started in 1997.

- The reef fish assemblage in the Cape Rodney to Okakari Point Marine Reserve continues to be distinct from that found in adjacent fished areas. In 2003 seven species were found to be primarily responsible for this: snapper *Pagrus auratus*, blue cod *Parapercis colias*, silver drummer *Kyphosus sydneyanus*, blue maomao *Scorpis violaceus*, banded wrasse *Notolabrus fucicola*, parore *Girella tricuspidata*, and eagle ray *Myliobatus tenuicaudatus*. The first four species were also on this list for previous years, while the last three were not.

- In autumn 2003, estimates made using Baited Underwater Video indicated that legal-sized snapper were 27.7 times more abundant inside the reserve than outside, an even greater difference than detected in previous autumn surveys. The increase was due to an approximate doubling in numbers of legal-sized fish within the reserve since autumn 2002. This was probably due to an exceptionally large influx of individuals from offshore waters. The increase in numbers, however, may be short-lived, as many of the additional fish can be expected to migrate out of the reserve again during the winter of 2003, and further surveys are required to determine the long-term effects on the numbers of "resident" snapper within the reserve.

- Spatial patterns of snapper abundance within the reserve were broadly similar to those found in earlier surveys, with densities declining toward the reserve boundary, although this trend was not as marked as in previous years. The absence of a plateau in the density gradient at the reserve centre suggests that the reserve may not be of sufficient size to realise a density level that is unaffected by fishing pressure at the boundaries. The absence of legal-sized snapper in the areas immediately outside the reserve suggests that fish leaving the reserve are rapidly removed by fishing.

Recommendations

- The fish monitoring programme should be continued with the current levels of sample replication regarded as a minimum level of effort.

- Seasonal variability is of sufficient magnitude in many of the more common species, and snapper in particular, that sampling on a six-monthly basis is desirable.

- More detailed stratification by habitat will help to determine whether large-scale habitat change inside the reserve has strong effects on the fish assemblage.

- The programme should be extended to include comparison with projected new reserves (e. g., Tiritiri Matangi) using identical sampling design and methodology. Timing of surveys must also be kept consistent between years. Comparison of this established reserve with a new reserve will provide insights as to 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, 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). 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 2000-2002 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 of snapper at Leigh from 1997-2002 found relative reserve density of legal-size fish (> 270 mm fork length) to vary between 7 and 90 times non-reserve density (Willis & Babcock 2000a), with marked seasonal variation in abundance, but a general indication that densities within the reserve were approaching a plateau.

This report presents the results of a survey conducted during autumn 2003 using identical techniques to previous years. The survey was intended to determine whether densities of snapper in particular had stabilised inside the reserve as suggested by previous recent surveys. If they had then this would allow the frequency of future

surveys to be reduced. This report should be read in conjunction with the previous reports (Willis & Babcock 2000a, Willis et al. 2003b).

Terminology/Abbreviations

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

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.

MDS: 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).

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

NPDisp: Computer programme used to test homogeneity of multivariate dispersions.

NPMANOVA: Non-parametric multivariate analysis of variance.

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).

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 x 5 m transects.

METHODS

Survey design

The 2003 census of the Cape Rodney to Okakari Point Marine Reserve was done from April 23-May 9 (BUV) and May 12-15 (UVC). Data for previous years were taken from Willis et al. (2003b).

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.
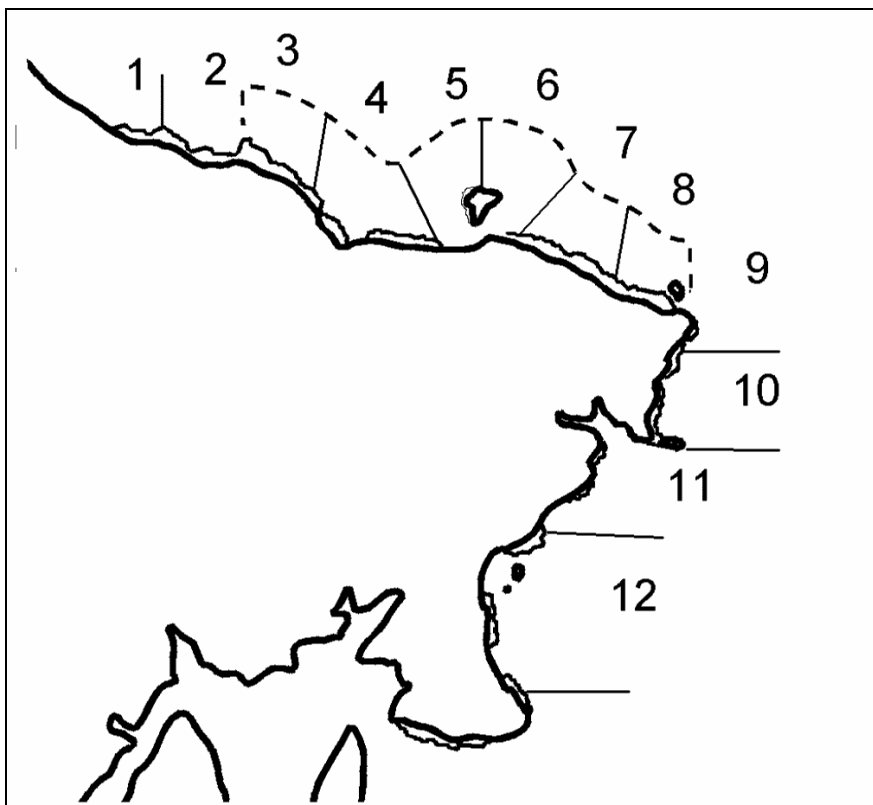

Figure 1. Map of sampling areas in and around the Cape Rodney to Okakari Point Marine Reserve. The dashed line shows the reserve boundary.

Survey methods

Underwater visual census

Within each site, three divers surveyed fishes within a total of nine 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

All BUV sampling was done from an anchored vessel (the University of Auckland's R. V. Hawere). The bait holder (Fig. 2) contained four whole pilchards (*Sardinops neopilchardus*) that were broken up to maximise the odour plume. External bait (one pilchard held in place by a cable tie) was placed on the lid of the bait holder. All baits were replaced for each replicate. Prior to deployment, location data (including GPS coordinates), depth, and the time were written down and filmed so that each video sequence was introduced by this information. The video unit was then lowered to the bottom while recording so that sequence timing could be taken from the time the unit made contact with the bottom. The rope and cable were buoyed at the surface to prevent them sinking into the field of view as the boat swung on its anchor. All sequences were of 30 min duration.
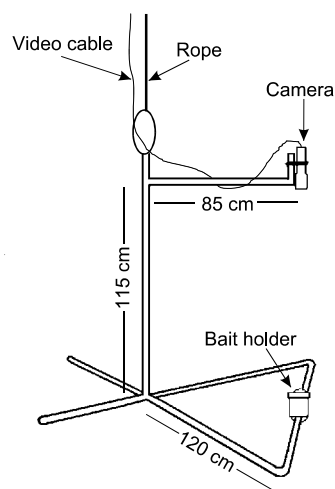

Figure 2. Baited underwater video assembly, with dimensions of the stand.

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 Mocha® 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

The variables of interest in the present study consist of 'count' data, which are discrete values rather than being continuous. As such, traditional linear models and tests (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 2003, 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). Quantile plots of residuals showed several outliers in the 2003 data and the Shapiro-Wilks test statistic indicated a significant departure from normality for each of the three variables ($P < 0.01$ in all cases). In addition, there was significant heterogeneity in the distribution of observations between the reserve and non-reserve samples for all snapper ($P < 0.01$) and for legal-sized snapper ($P < 0.001$).

Ratios of densities of snapper between reserve and non-reserve areas for BUV data were therefore assessed using generalised linear models (GLMs, 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 computer package (Ihaka and Gentleman 1996). 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 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

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

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. 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 ($\times 2$) 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 last year's census (autumn 2002) with those obtained in the current year (autumn 2003). 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. All non-parametric tests were done using the R statistical computer program (Ihaka and Gentleman 1996).

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* = 9 transects were summed for each variable to obtain a single observation for each station). There were 35 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, 4, 5, 6, 7 and 8) and 6 areas located outside the reserve (areas 1, 2, 9, 10, 11 and 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 non-parametric multivariate analysis of variance (NPMANOVA, Anderson 2001a), with "Status" (reserve versus non-reserve) treated as a fixed factor and "Areas" treated as a random factor, nested within "Status". For appropriate tests of individual terms, restricted permutations were used (e. g., see Anderson 2001b). Data were also tested for homogeneity of multivariate dispersions using the computer programme NPDisp (Anderson 2000). 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 in press). 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 (NPMANOVA, NPDisp and CAP) were obtained using 4999 permutations. For further details concerning any of the multivariate methods used in the present investigation, see Appendix 1 of Willis et al. (2003b).

Non-metric multidimensional scaling (MDS) was used to display long-term changes at the community level. MDS 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 MDS 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.

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 ± 2.5 individuals per BUV drop (paired Wilcoxon signed rank test comparing autumn

2002 with autumn 2003: $P = 0.036$) (Fig. 3a, Table 1). Outside the reserve the mean total snapper density in 2003 was only 4.1 ± 1.5 individuals per BUV drop, a value that differed little from previous years (autumn densities ranged from ~3.5-6.7 during 1998-2002). The increase within the reserve for 2003 was wholly attributable to legal (> 270 mm fork length) individuals, numbers of which more than doubled from a mean density of $10.3 \pm \sim 1.2$ individuals per BUV drop in 2002 to 21.9 ± 2.9 in 2003 (paired Wilcoxon signed rank test: $P = 0.031$) (Fig. 3b, Table 1). Legal-size snapper are now estimated to be 27.7 times more abundant inside the reserve than outside, an increase over the ratios of 10.4 for autumn 2001 and 13.1 for autumn 2002 (Table 1). Densities of undersize fish were consistent with previous years (4.8 ± 0.8 individuals per BUV drop inside the reserve and 3.3 ± 1.3 outside), and continued the trend of no consistent difference between reserve and non-reserve areas (Fig. 3c).

The spatial distribution pattern of legal snapper was 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 toward the reserve boundaries was much less pronounced in 2003 than in previous years, with boundary areas 3 and 8 containing 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. W., 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 2003 was over 100 mm greater than that of fish outside the reserve (Fig. 5, Table 2). Comparisons of the sizes of legal-size fish were significant for only two of the five survey dates (Table 2). The reason for this is apparent in Fig. 5, where it can be seen that only one snapper larger than 450 mm fork length was seen outside the reserve (in the 2003 survey), but fish larger than this were common within the reserve. The non-significant results were mainly due to very low sample sizes of legal snapper outside the reserve, giving very low statistical power for the comparisons.

Comparisons of size data from the three autumn surveys reveals an increase over time in the average sizes of snapper from both inside and outside the marine reserve. The average size of all snapper inside the reserve increased by 21.1 mm from autumn 2001 to autumn 2002, and a further 22.8 mm from autumn 2002 to autumn 2003 (Table 2). The respective increases in the average size from year to year for fish outside the reserve were 10.9 and 27.7 mm.

Table 1. Mean densities of snapper *Pagrus auratus* inside and outside the Cape
Rodney to Okakari Point Marine Reserve, from 2000-2003 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

Table 2. Mean sizes of snapper *Pagrus auratus* inside and outside the Cape Rodney to Okakari Point Marine Reserve, from 2000-2003 BUV surveys. Statistically significant ($P < 0.05$) differences are denoted by *. N = number of fish.

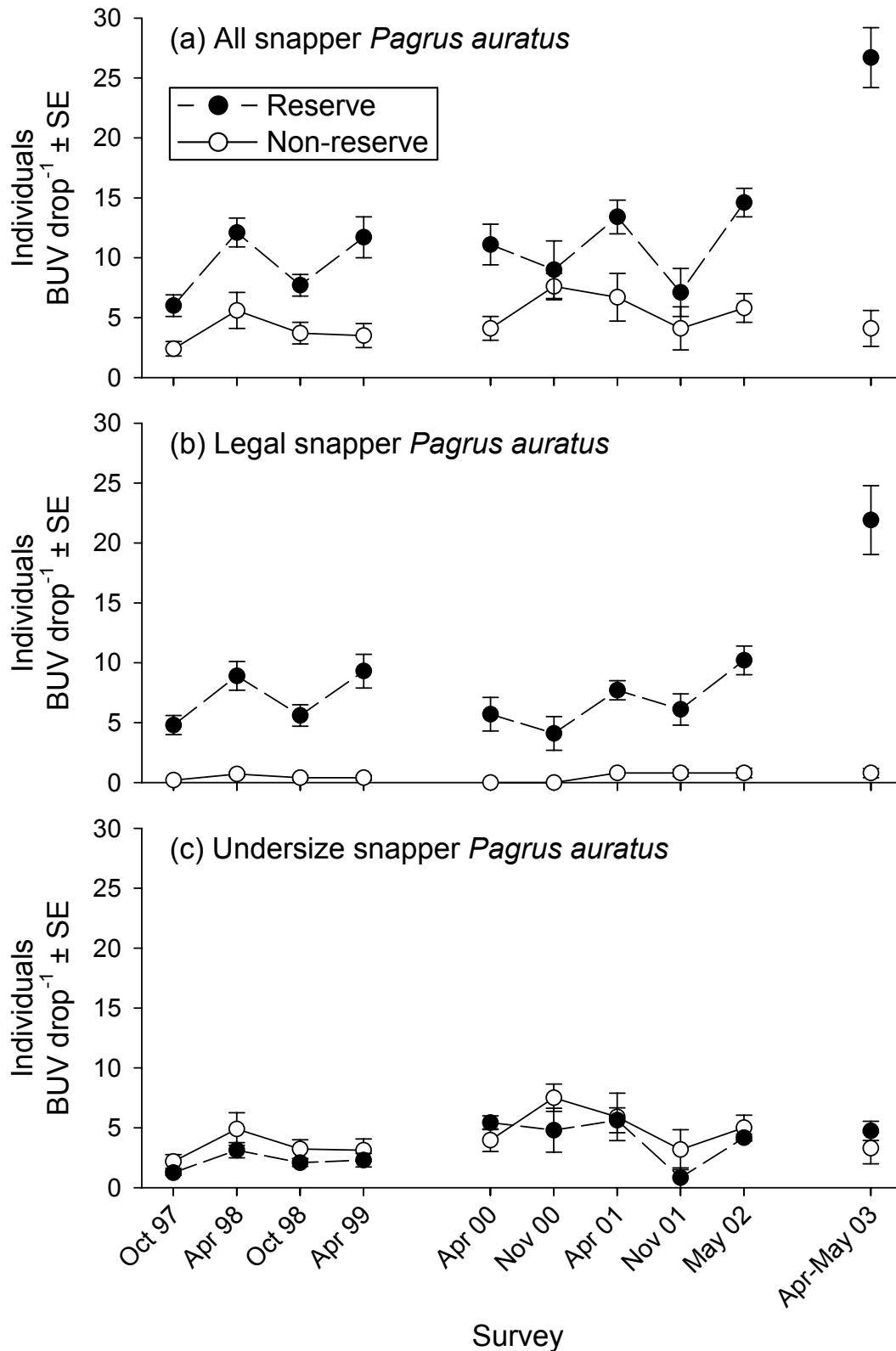| Survey	Reserve mean fork length (mm)	N: Reserve	Non-reserve mean fork length (mm)	N: Non-reserve	Difference between means (mm)	95% CI
All snapper
Spring 2000	288.9	197	148.8	159	140.2*	24.9
Autumn 2001	307.7	322	203.5	160	104.1*	18.8
Spring 2001	389.2	165	217.9	94	171.3*	25.4
Autumn 2002	328.8	342	214.4	135	114.4*	19.1
Autumn 2003	351.6	640	242.1	98	109.5*	20.1
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

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).
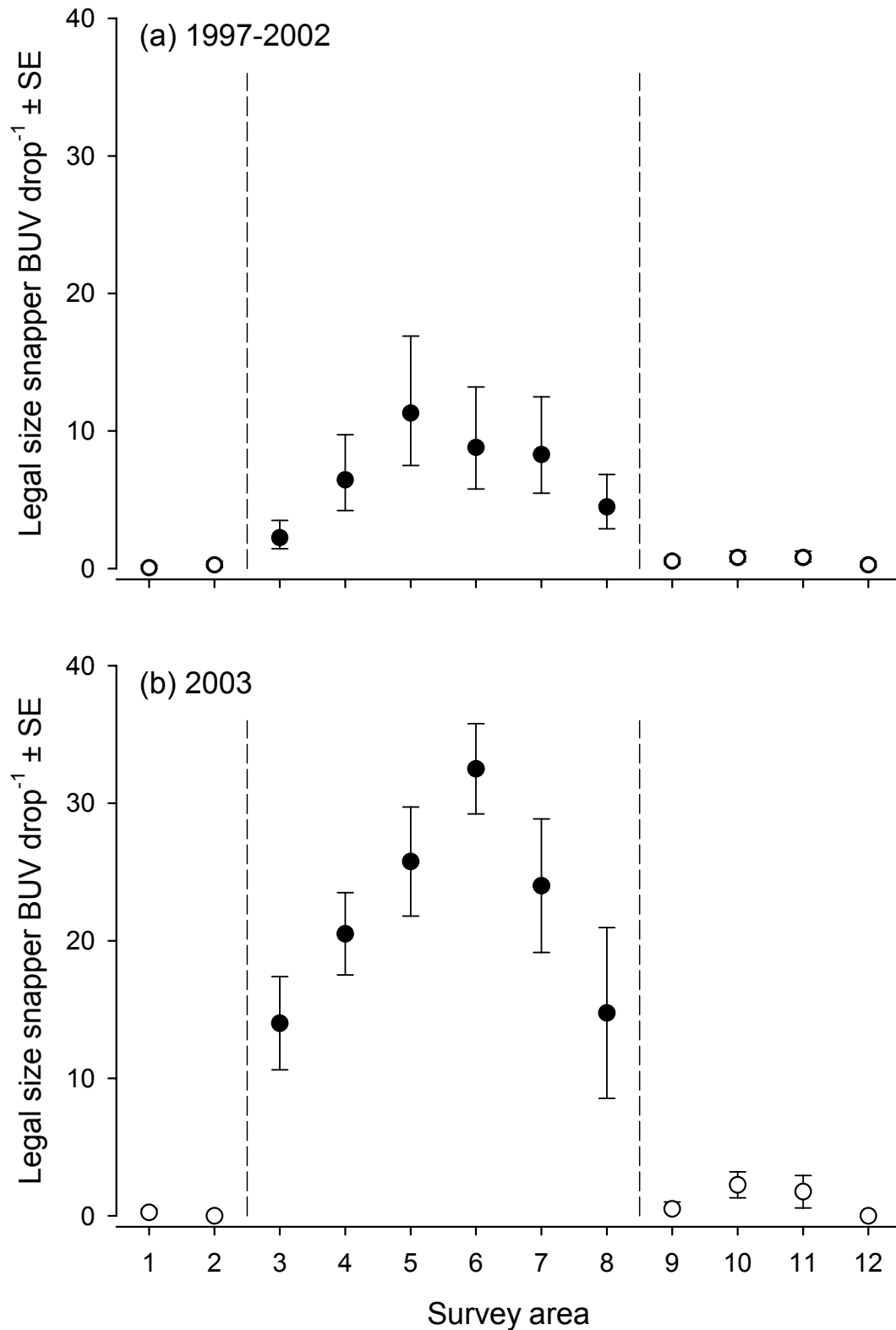
Figure 4. Expected relative density of legal-size snapper *Pagrus auratus* within the twelve survey areas, based on (a) modelled data from nine BUV surveys (October 1997 – May 2002), and (b) 2003 BUV data. Closed symbols are within the reserve, open symbols are fished areas. Dashed vertical lines indicate reserve boundaries.
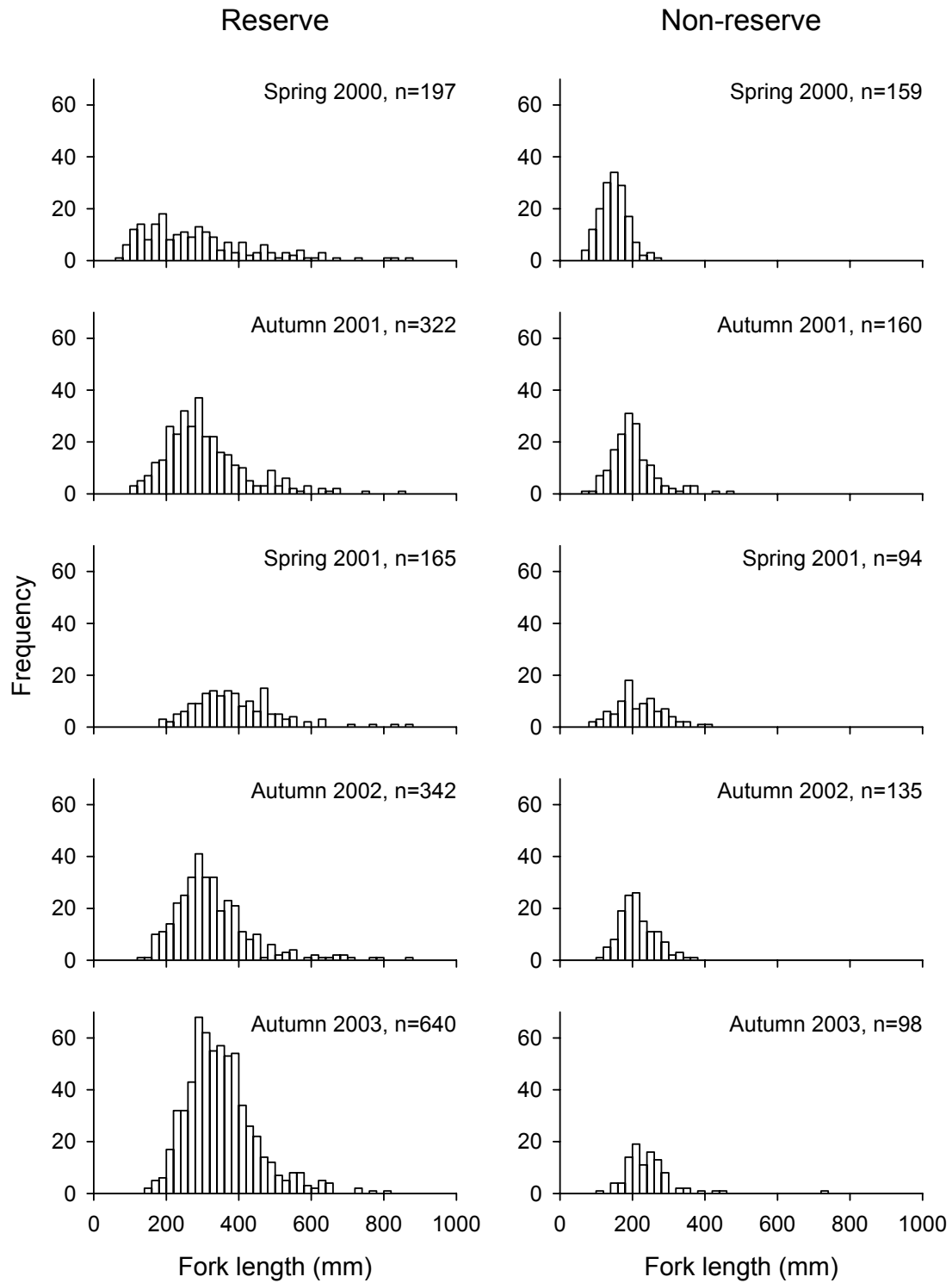
Figure 5. Size frequency distributions of snapper *Pagrus auratus* inside and outside the Cape Rodney to Okakari Point Marine Reserve from 2000-2003, as measured using BUV.

Blue cod Parapercis colias

Blue cod densities within the marine reserve declined steeply from ~1.8 individuals per BUV drop in spring 1997 to only ~0.2 in autumn 1999, and then fluctuated around 0.5 until the autumn 2002 survey (Fig. 6a, Table 3). The autumn 2003 survey revealed a ~1.6-fold recovery in numbers to 0.79 ± 0.25 individuals per BUV drop, although this increase was not statistically significant (paired Wilcoxon signed rank test: P = 0.104). No blue cod were recorded outside of the marine reserve in autumn 2003. Willis et al. (2003b) suggested that the decline in cod density might be attributable to warmer than average sea surface temperatures that occurred from the winter of 1998 until the end of 1999 (Fig. 6b). A much longer time-series would be required to test the hypothesis that there is a correlation of blue cod densities with sea surface temperature.

Blue cod within the reserve were generally larger than those that occurred outside (Table 4), but numbers were too low for meaningful statistical analysis. No comparison could be made for autumn 2003, as no blue cod were recorded outside of the marine reserve during that survey.

Table 3. Mean densities of blue cod *Parapercis colias* inside and outside the Cape Rodney to Okakari Point Marine Reserve, from 2000-2003 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	∞*		

Table 4. Mean sizes of blue cod *Parapercis colias* inside and outside the Cape Rodney to Okakari Point Marine Reserve, from 2000-2003 BUV surveys. Statistically significant (P < 0.05) differences are denoted by *. N = number of fish.

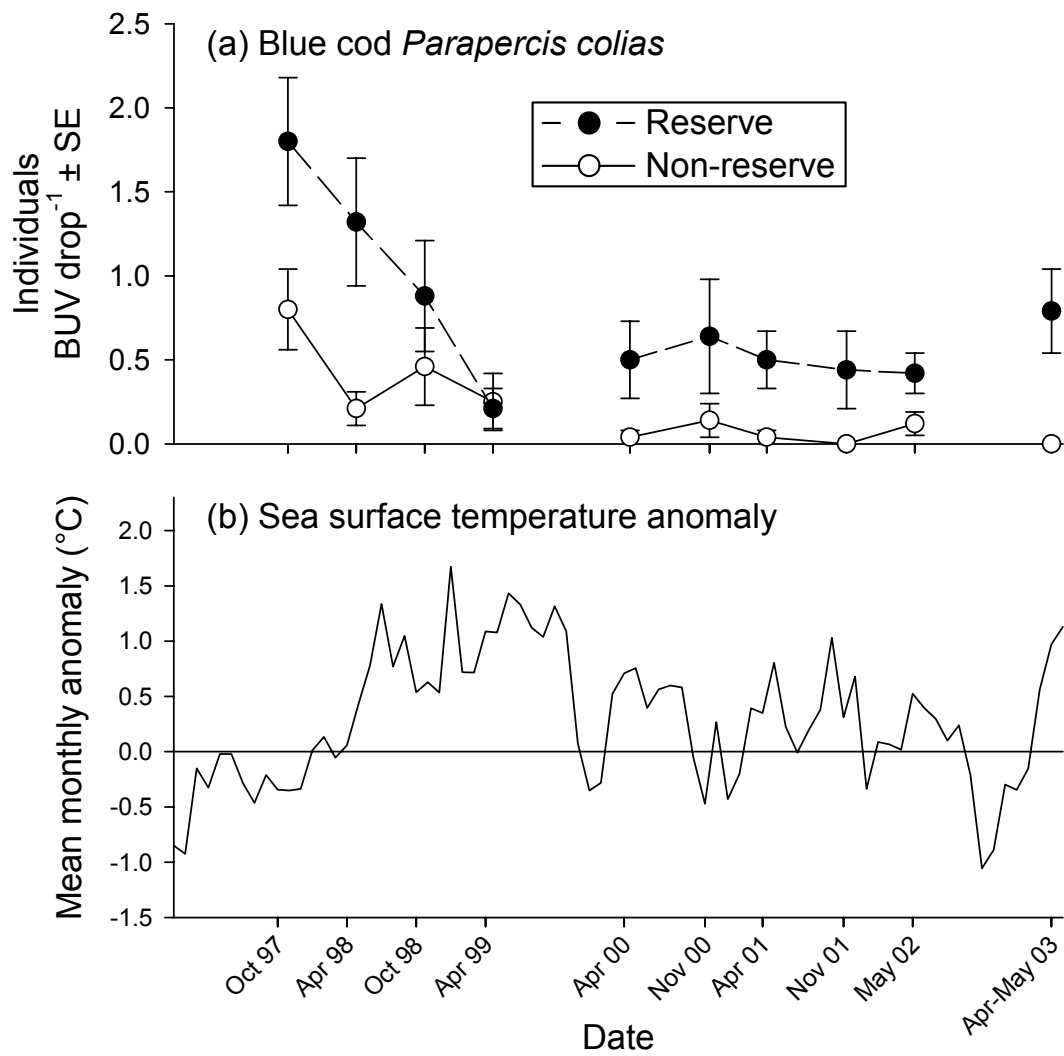| Survey	Reserve mean fork length (mm)	N: Reserve	Non-reserve mean fork length (mm)	N: Non-reserve	Difference between means (mm)	95% CI
Spring 2000	314.0	14	242.7	4	71.2	75.8
Autumn 2001	257.2	12	117.0	1	140.2	-
Spring 2001	282.9	11	-	0	-	-
Autumn 2002	257.6	11	197.7	3	60.0	66.6
Autumn 2003	322.9	19	-	0	-	-

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 1966-96).

Underwater visual census

Community-level patterns

There was a significant difference between fish assemblages from areas outside the marine reserve compared to those inside the marine reserve (i. e., a significant effect of "Status" in the NPMANOVA, Table 5). In addition, the average percentage difference among fish assemblages from the non-reserve stations (i. e., 52.3%) was large compared to that from the reserve stations (42.4%). This suggested that assemblages at non-reserve sites might be more variable than those inside the reserve. However, it was found that there was in fact no significant difference in the multivariate dispersion of fish assemblages inside versus outside the reserve (NPDisp, Table 6). The apparently large average dispersion in the non-reserve sites was actually caused by a single outlier at station number 10 from area 1 outside the reserve. The habitat at this station consisted of a sandy bottom and here, no fish were actually observed in the entire set of 9 transects surveyed by divers. The Bray-Curtis dissimilarity between this station and every other station is 100%, as they have no species in common, making this station a gross outlier. When this outlier was removed from the analysis, the average percentage difference among fish assemblages from the non-reserve stations was 42.7%, very similar to that for the reserves. Thus, it became clear that differences between reserve and non-reserve sites were due to location effects (i. e., specific differences in composition or relative abundances of species), not differences in variability.

The unconstrained PCO plot also suggested a difference between the fish assemblages inside versus outside the reserve, although there was some overlap (Fig. 7a). That is, the symbols on the plot were not well-mixed (reserve sites appeared more in the upper left, while non-reserve sites appeared more in the lower right), but neither did they separate cleanly in two dimensions (Fig. 7a). In addition, the variation among stations within the same area was in many cases as large as the variation among different areas (i. e., assemblages from the same area tended to be as far away from each other on the plot as assemblages from different areas, Fig. 7b). Thus, it is not surprising that the factor "Areas" was not statistically significant in the NPMANOVA (Table 5). 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 12). A much clearer distinction between the fish assemblages from reserve versus non-reserve stations is seen in the constrained (CAP) plot (Fig. 8). These two groups were found to be quite distinct, with very little overlap and a calculated misclassification error of only 13%.

Several fish species showed correlations of $|r| > 0.20$ with the canonical axis (Table 7). Those with a positive correlation are expected to occur with greater frequency and/or abundance in reserve sites, while those with a negative correlation are expected to occur with greater frequency and/or abundance in non-reserve sites. Fish associated with reserve sites were: snapper (*Pagrus auratus*), silver drummer (*Kyphosus sydneyanus*), blue cod (*Parapercis colias*), blue maomao (*Scorpis violaceus*), banded wrasse (*Notolabrus fucicola*), parore (*Girella tricuspidata*), and eagle ray (*Myliobatus*

tenuicaudatus). The first four species were also on the same list in previous years, while the last three were not and replaced demoiselle (*Chromis dispilus*), trevally (*Pseudocaranx dentex*), and butterfish (*Odax pullus*) (see Table 10 in Willis et al. 2003b). Fish associated with non-reserve sites were: goatfish (*Upeneichthys lineatus*), bigeye (*Pempheris adspersa*), sweep (*Scorpis lineolatus*), spotty (*Notolabrus celidotus*) and hiwihiwi (*Chironemus marmoratus*).

Long-term changes at the community level are shown in a non-metric multidimensional scaling (MDS) plot (Fig. 9). 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.

Table 5. NPMANOVA on the basis of the Bray-Curtis dissimilarity for ln(y+1) transformed species abundance data (35 species). The *P*-value for Status was obtained using permutation of whole area units, while the *P*-value for Areas was obtained using permutation of individual stations but restricted within each level of Status.

Source	df	SS	MS	*F*	*P*
Status	1	2502.69	2502.69	1.947	0.0284
Areas(Status)	10	12856.73	1285.67	0.974	0.5574
Residual	12	15840.64	1320.05		
Total	23	31200.05			

Table 6. NPDisp on the basis of the Bray-Curtis dissimilarity for ln(y+1) transformed species abundance data (35 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 sites, as shown below. The *P*-value was obtained as described in Table 1.

Source	df	SS	MS	*F*	*P*
Status	1	64.90	64.90	0.297	0.6904
Areas(Status)	10	2183.83	218.38	---	---
Residual	12	0.00	0.00		
Total	23	2248.73			

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 15% of the samples.

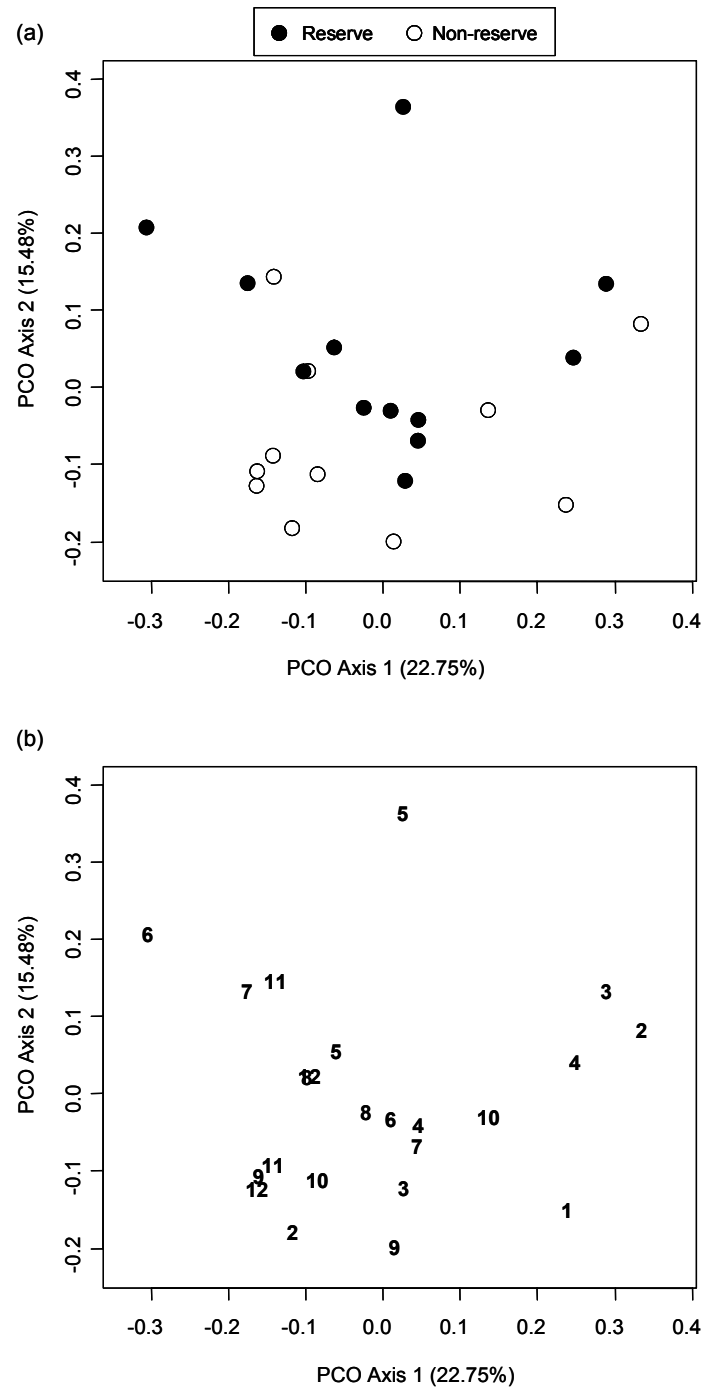| Positive correlation (reserve)		*r*
Snapper	*Pagrus auratus*	0.615
Silver drummer	*Kyphosus sydneyanus*	0.486
Blue cod	*Parapercis colias*	0.457
Parore	*Girella tricuspidata*	0.379
Blue maomao	*Scorpis violaceus*	0.317
Eagle ray	*Myliobatus tenuicaudatus*	0.267
Banded wrasse	*Notolabrus fucicola*	0.227
Negative correlation (non-reserve)		
Goatfish	*Upeneichthys lineatus*	-0.396
Bigeye	*Pempheris adspersa*	-0.327
Sweep	*Scorpis lineolatus*	-0.306
Spotty	*Notolabrus celidotus*	-0.260
Hiwihiwi	*Chironemus marmoratus*	-0.191

Fig. 7. Ordination plot of the first two PCO axes (explaining 38.23% of the original variability) based on Bray-Curtis dissimilarities of ln(y+1) transformed species abundance data (35 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). Note that 1 station from area 1 was not included in these plots as it was an outlier and included no fish (see text for details).
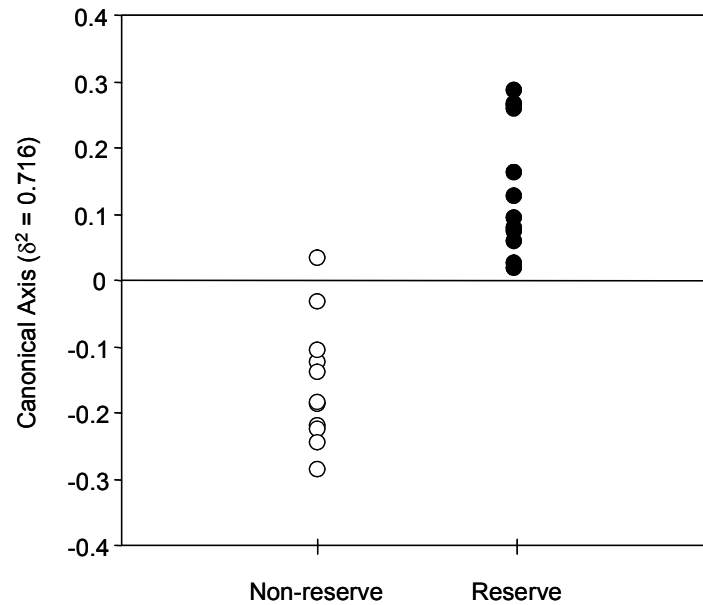
Fig. 8. Plot of the canonical axis from a CAP constrained ordination to discriminate fish assemblages from reserve versus non-reserve stations. The discriminant analysis was done on the first $m = 7$ PCO axes (which explained 93.67% of the original variability) from Bray-Curtis dissimilarities of $\ln(y+1)$ transformed species abundances (35 species).
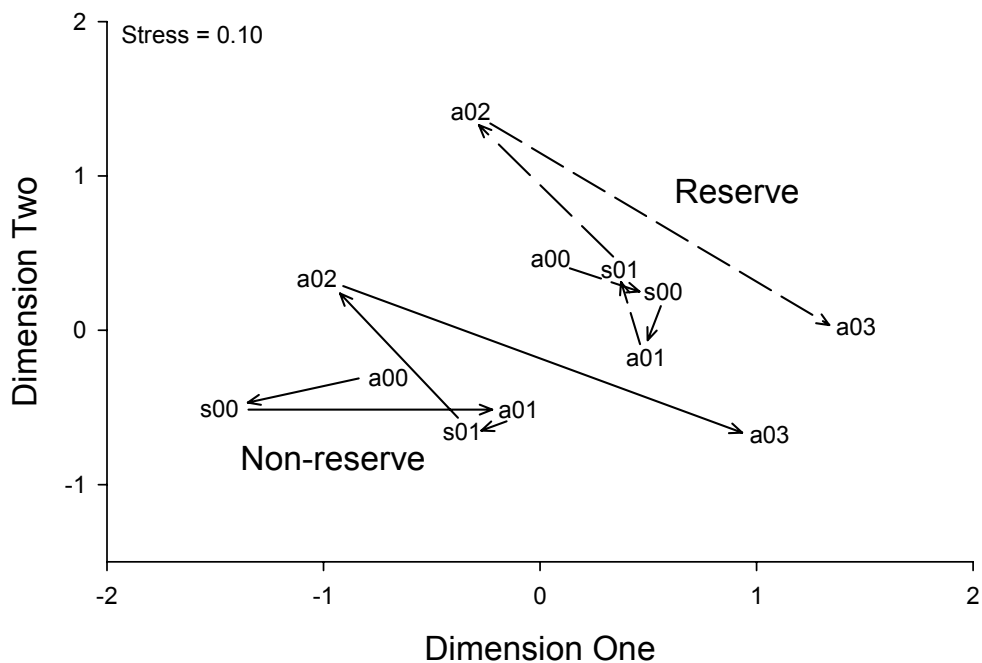

Fig. 9. Non-metric multidimensional scaling (MDS) plot of changes in the fish communities inside and outside the Cape Rodney to Okakari Point Marine Reserve. Pelagic species were excluded from the analysis. Codes correspond to season and year, e. g., a00 = autumn 2000, s00 = spring 2000.

Individual species

Individual species continued to vary considerably in their long-term abundances as estimated using UVC. In autumn 2003 the UVC estimate of the mean density of snapper within the reserve was 4.1-fold higher than the previous year's value (0.98 vs 3.97 individuals per 125 m^2; paired Wilcoxon signed rank test: P = 0.031) (Fig. 10), reflecting the pattern seen in the recent BUV data (Fig. 3a). In 2003, snapper densities were 4.0-fold higher inside the reserve than outside (3.97 vs 0.99 individuals per 125 m^2; two-sample Wilcoxon rank sum test: P = 0.030). Blue cod displayed similar patterns to snapper, with mean densities in the reserve for 2003 increasing 4.2-fold over the previous year (from 0.11 to 0.46 individuals per 125 m^2), although this increase was not detected as statistically significant (paired Wilcoxon signed rank test: P = 0.281, Fig. 10). In 2003 blue cod densities were 5.8-fold higher inside the reserve than outside (0.46 vs 0.08 individuals per 125 m^2), which was a statistically significant difference (two-sample Wilcoxon rank sum test: P = 0.028). Red moki densities showed high variability in 2003 (see the relative sizes of error bars for 2003 compared to previous years in Fig. 10), with mean densities increasing both inside and outside the reserve.

The relative densities of spotty and banded wrasse followed very similar patterns in time (Fig. 11), which was probably the result of similar recruitment success in the summer of 2000-01. Densities were generally higher outside the reserve. The relative density of trevally was variable, as expected for a pelagic, schooling species, but usually higher inside the reserve (Fig. 11). The zero values recorded for autumn 2003 are a function of the patchiness of this schooling species – at least a hundred individuals were seen near Shag Rock at Goat Island in July 2003 (R. B. T., pers. obs.).

Leatherjackets and goatfish exhibited strong seasonality in abundance, which was primarily due to recruitment events (Fig. 12), but there were no consistent differences between reserve and non-reserve average densities. Silver drummer were usually only seen inside the reserve (Fig. 12) – usually in Alphabet Bay (Area 5) or the vicinity of Tabletop Reef (Area 7). In 2003, 22 individuals were seen outside the reserve at Cape Rodney (Area 9).

The density of sweep was less variable than expected for a schooling species (Fig. 13), and followed similar trends through time inside and outside the reserve. Blue maomao were variable (they were either absent or abundant), but were found in greater densities, on average, inside the reserve, as were demoiselles (Fig. 13).

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

Butterfish have generally been more abundant in the reserve than in fished areas (Fig. 15), although densities were low. This species responds negatively to divers, so its abundance was probably underestimated.
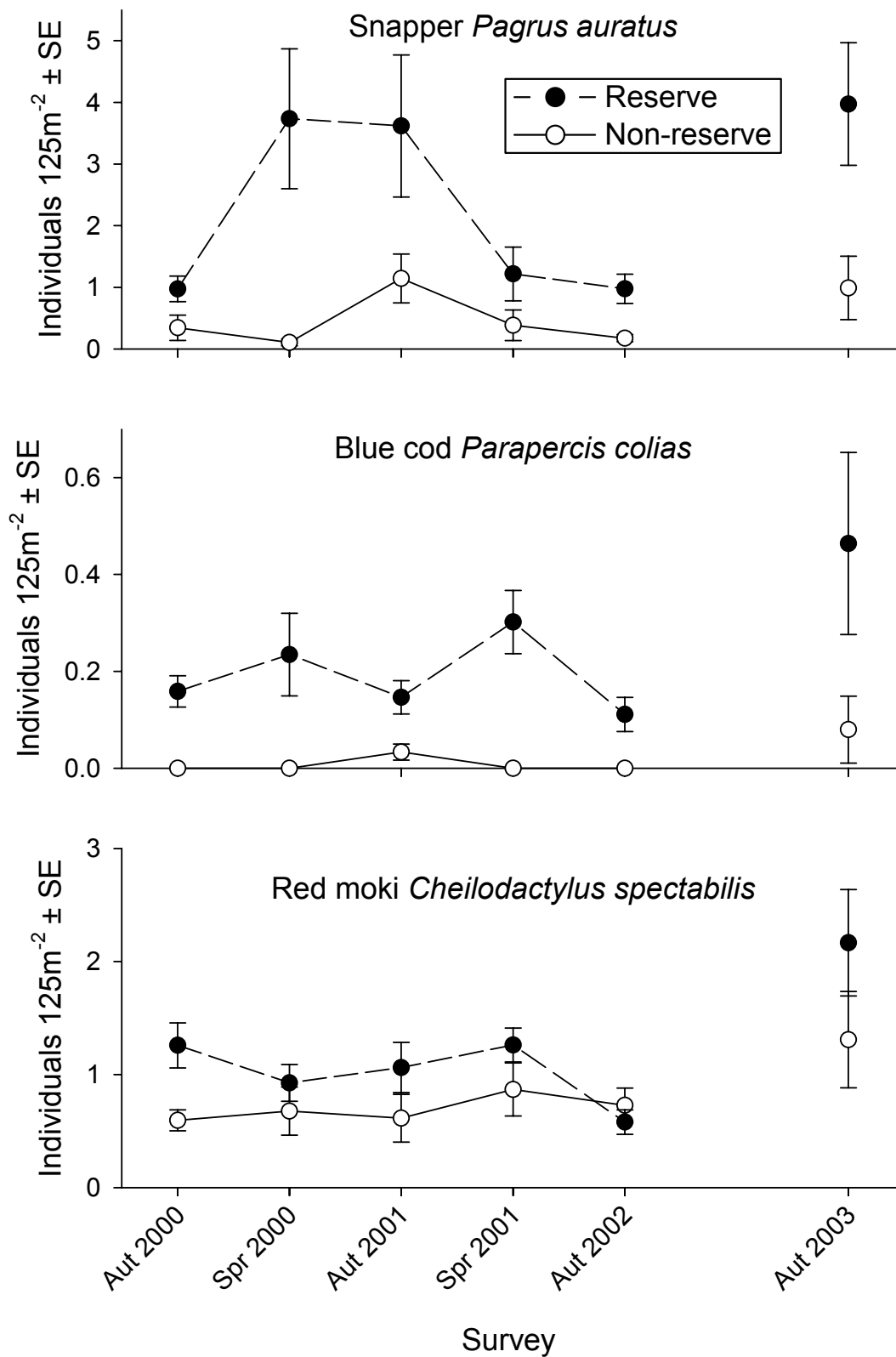
Figure 10. Long term trends in the densities of snapper, blue cod, and red moki inside and outside the Cape Rodney to Okakari Point Marine Reserve, as measured using UVC.
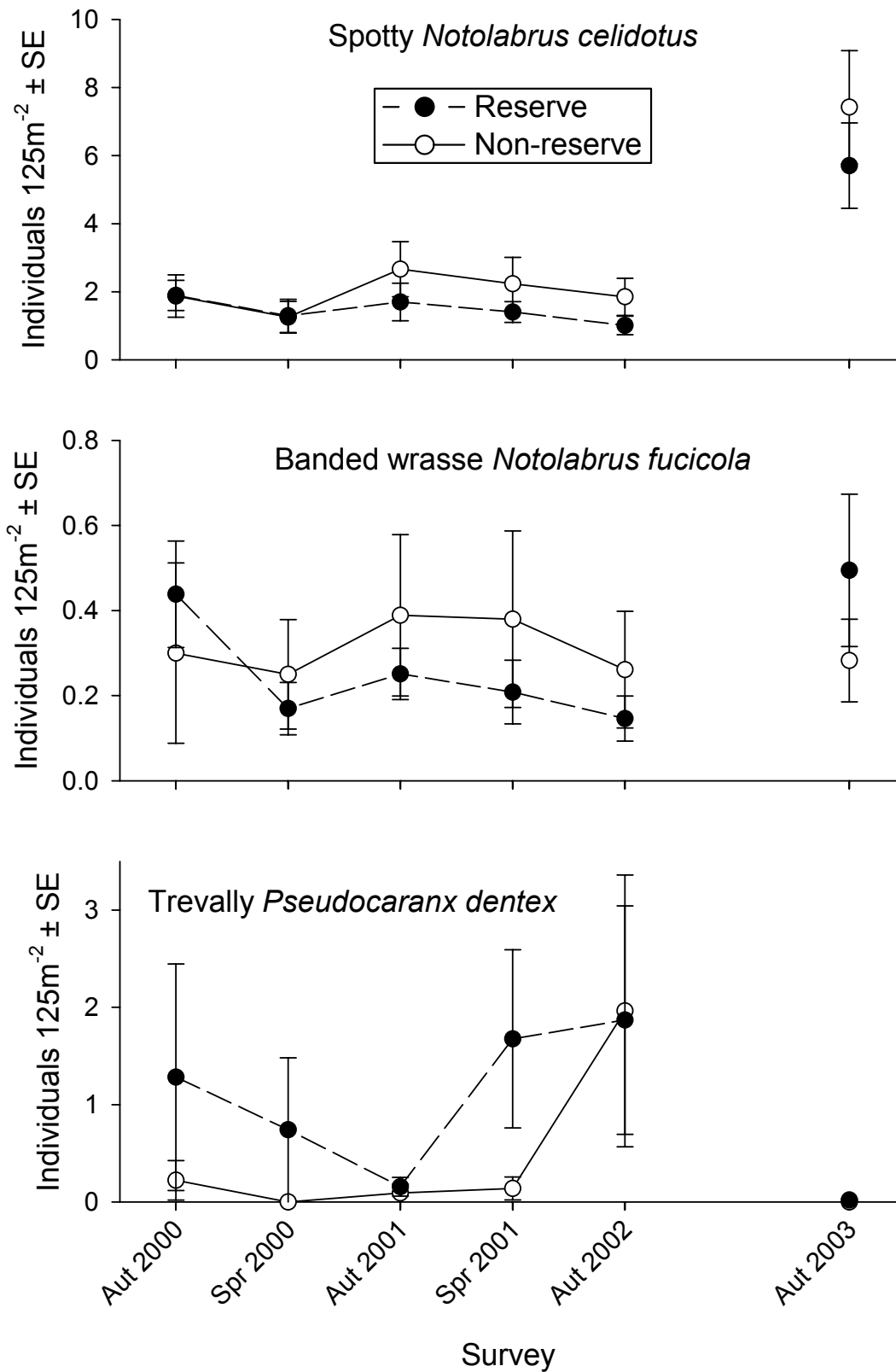
Figure 11. Long term trends in the densities of spotty, banded wrasse, and trevally inside and outside the Cape Rodney to Okakari Point Marine Reserve, as measured using UVC.
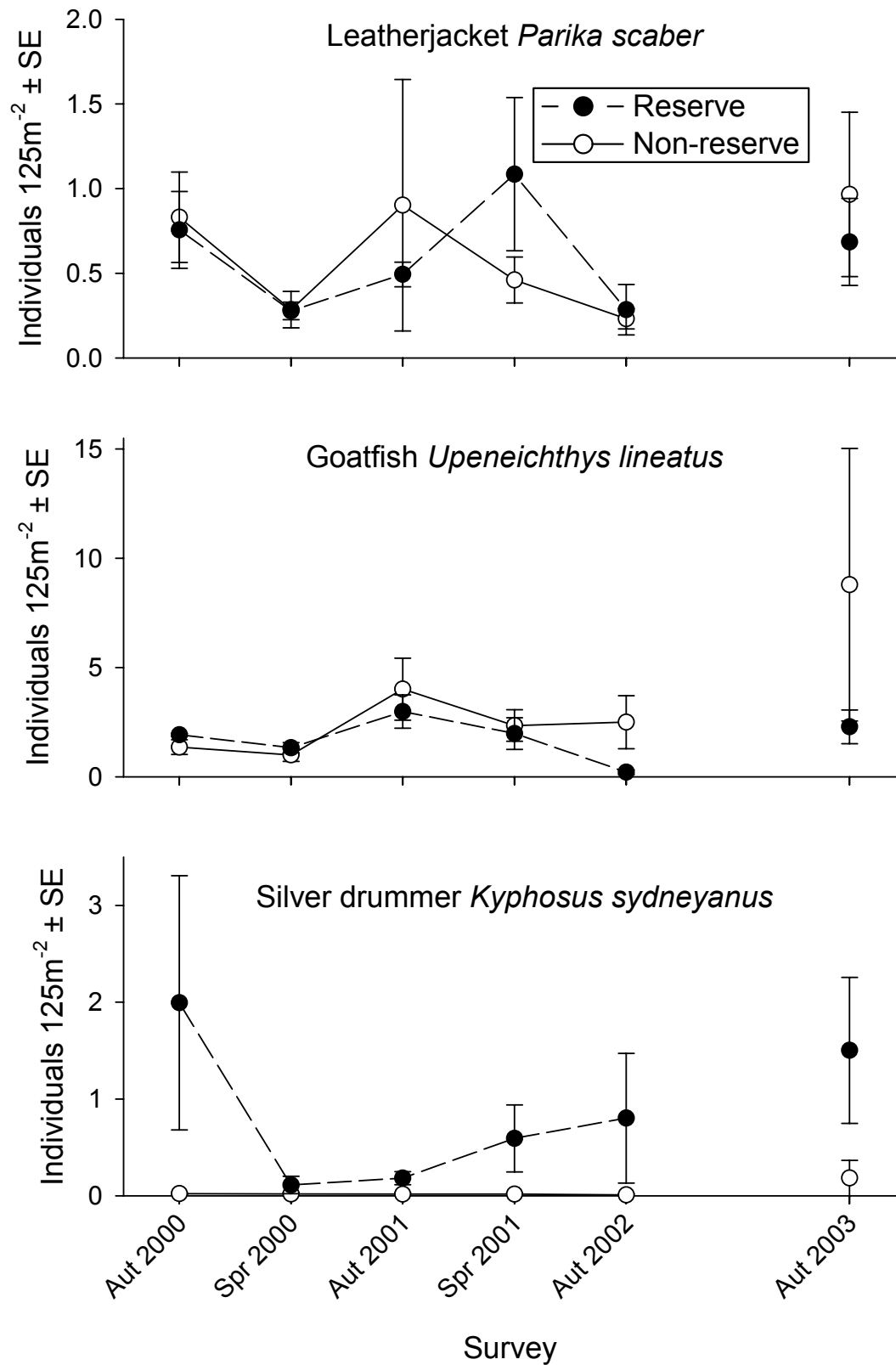
Figure 12. Long term trends in the densities of leatherjacket, goatfish, and silver drummer inside and outside the Cape Rodney to Okakari Point Marine Reserve, as measured using UVC.
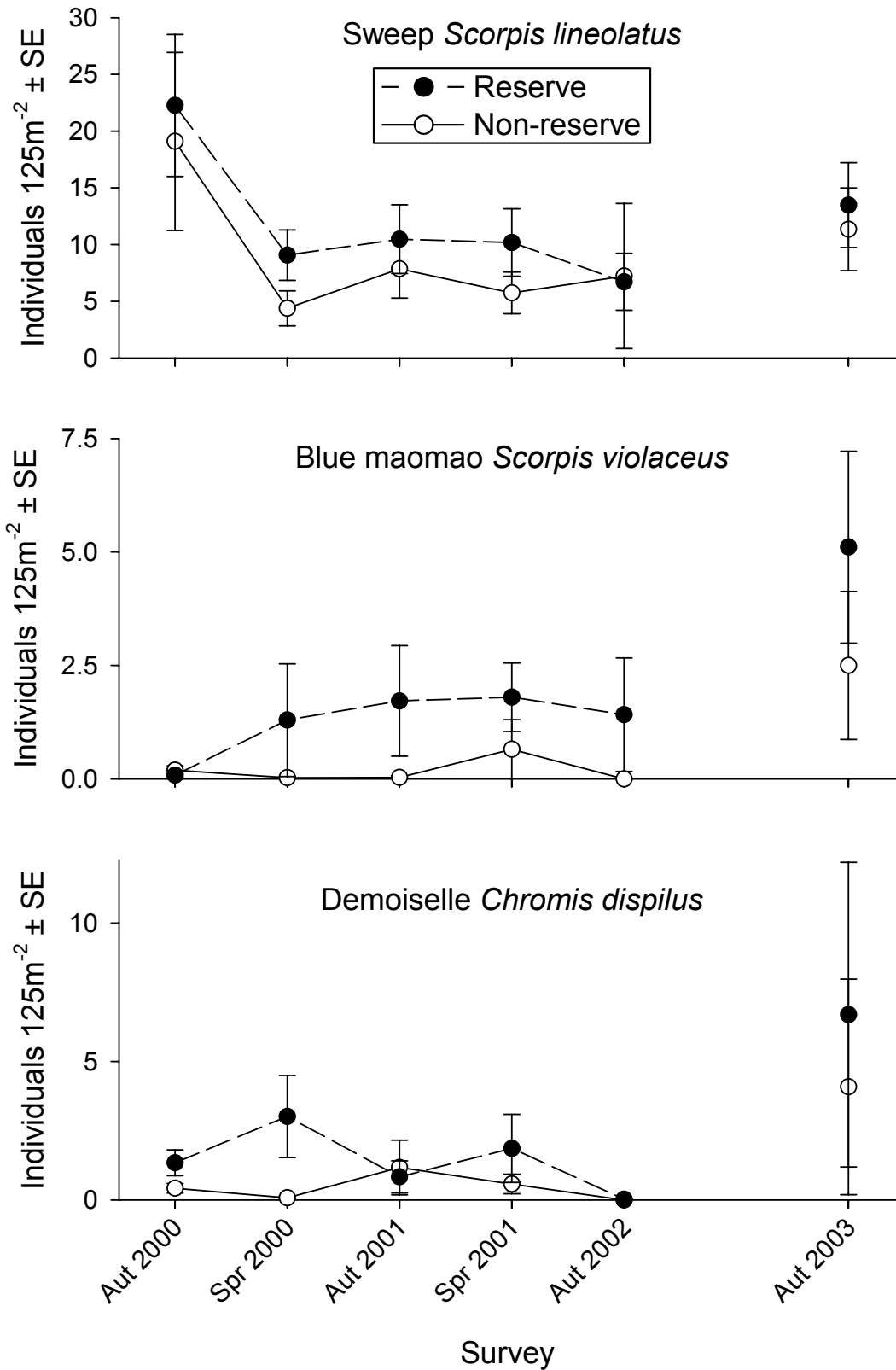
Figure 13. Long term trends in the densities of sweep, blue maomao, and demoiselle inside and outside the Cape Rodney to Okakari Point Marine Reserve, as measured using UVC.
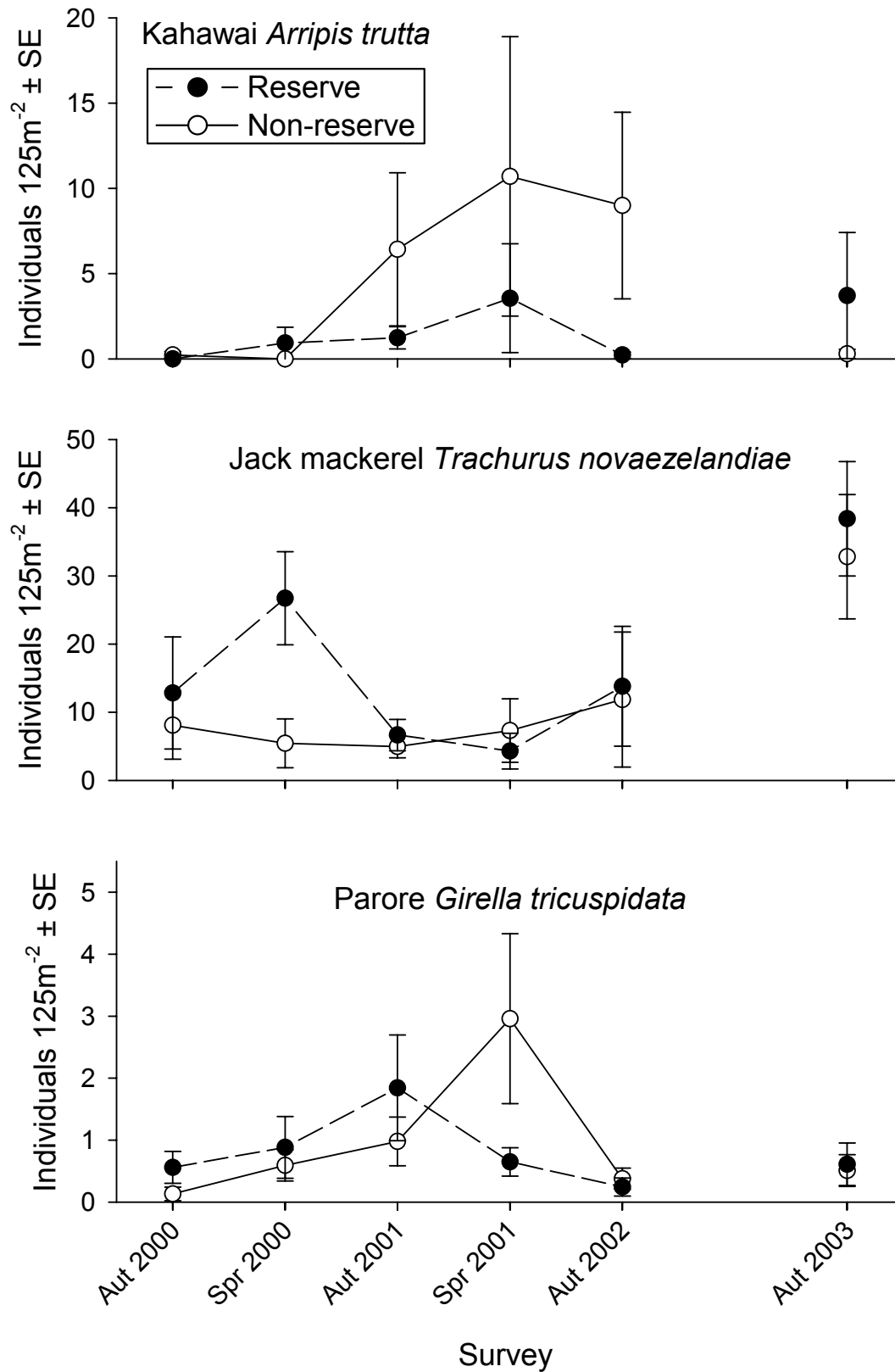
Figure 14. Long term trends in the densities of kahawai, jack mackerel, and parore inside and outside the Cape Rodney to Okakari Point Marine Reserve, as measured using UVC.
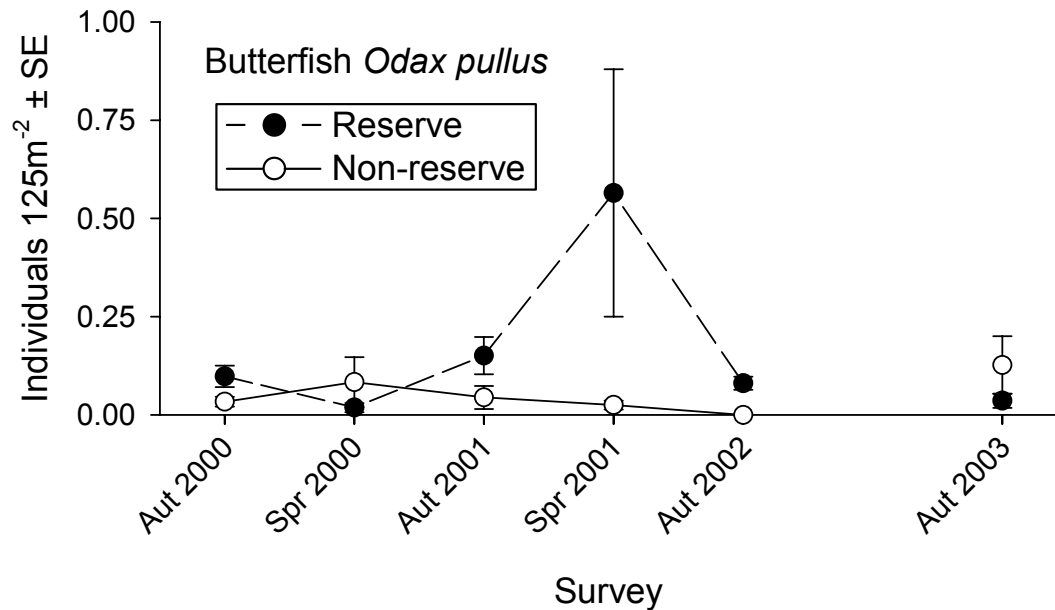
Figure 15. Long term trends in the densities of butterfish inside and outside the Cape Rodney to Okakari Point Marine Reserve, as measured using UVC.

DISCUSSION

As for past surveys, the autumn 2003 survey upon which this report is based revealed major differences between fish communities living in the Cape Rodney to Okakari Point Marine Reserve and 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*.

In autumn 2003, estimates made using BUV indicated that numbers of legal-sized snapper (> 270 mm fork length) were about 28 times higher in reserve than non-reserve areas. This difference was much greater than for the previous autumn surveys, due to an apparent doubling in numbers of legal-sized snapper in the reserve since autumn 2002. Over the preceding three autumn surveys there was the suggestion of a gradual increase in numbers of legal snapper in the reserve, but the 2003 densities are well above this trendline (Fig. 3b). The increase in 2003 was not due to methodological changes associated with the departure of key staff (T. J. Willis) prior to the 2003 survey. N. Usmar was heavily involved in video analyses in both the 2002 and 2003 surveys, and D. Egli was experienced in using the BUV prior to running the 2003 survey. Nor was the increase likely due to differences in timing of the surveys: the autumn 2002 BUV survey was conducted from May 3-11 (except for two non-reserve sites in early June), while the 2003 survey was done from April 23-May 9. The increase in numbers was not due to the influx of a single cohort of young fish; the

"additional"[1] fish spanned the full size range and included many individuals larger than 450 mm fork length (Fig. 16). These large individuals are unlikely to have come from immediately outside the reserve (Fig. 5), leaving an exceptionally large immigration of fish from more distant offshore waters as the most likely explanation for the increase. This is consistent with past research, which suggests that the snapper population in the reserve is comprised of (1) "resident" fish that are present all year round and dominate the spring surveys, and (2) migratory fish that move in and out of the reserve seasonally (possibly related to spawning) and are most strongly represented in the autumn surveys (Egli and Babcock 2002, Willis et al. 2001, 2003a). Another possibility is that the attractiveness of the BUV to the resident snapper was higher in 2003 than 2002 (this might happen if, for example, if the fish became more accustomed to the BUV or were particularly hungry due to a natural food shortage). However, the increase in snapper densities from 2002 to 2003 evident in the independently gathered UVC data (Fig. 10) provides reassurance that the increase in densities is not an artefact of the BUV. Assuming the exceptional numbers of legal snapper recorded in 2003 are real, it is impossible to predict whether they represent (1) an aberration in a stable long-term pattern, (2) a new plateau, or (3) the beginning or acceleration of an upward trend. Patterns from past years suggest that a large proportion of the fish detected in the autumn survey will leave the reserve over the winter. It would be interesting to know how many individuals remain in the reserve this coming spring to become "residents". These possibilities can only be distinguished by further surveys, which would clearly be more robust if based on both BUV and UVC. It would also be informative to run surveys in both spring and autumn, as the spring survey gives a better indication of the density of permanent residents (which is a "true" indicator of reserve effectiveness; Willis et al. 2003b), while the autumn survey quantifies what is probably closer to the maximum number of fish present at any time of the year (which is important for estimating their ecological impact within the reserve). If the increase is real then we are unaware of any environmental change that may have triggered it. In the six months or so leading up to the 2003 survey the sea surface temperature increased rapidly relative to the long-term mean (Fig. 6b), but was still well within the range of previous years in which there was no large increase in snapper numbers.

Within-reserve patterns of snapper density were similar to those found in previous surveys, lending further support to the idea that vulnerability to fishing increases near the boundaries. High fishing pressure at the boundaries appears to reduce the effective size of the reserve slightly, although the boundary areas still contain higher densities than any fished area. In 2003 the decrease in densities from the centre to the outermost reserve areas was far less dramatic than in previous years. One possible explanation for this is that the higher overall snapper densities in the reserve caused the displacement of individual fish toward the boundaries. Another is that the elevated densities were better able to compensate for the loss of fishes caught when they move outside the reserve. In 2003 the differences in snapper numbers between areas on opposing sides of each boundary were particularly strong, suggesting that any fish leaving the reserve are very rapidly removed by fishing.

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[1] We put "additional" in quotes because the difference between fish populations present in the reserve in autumn 2002 and autumn 2003 is not due solely to new immigrants (except in the unlikely case that all of the individuals leaving before spring 2002 returned again before the autumn 2003 survey).

BUV and UVC both revealed increases in average blue cod densities within the marine reserve compared to previous years, although this pattern was not statistically significant (Figs 6a, 10). It remains to be seen whether this represents the beginning of a recovery to higher numbers.

Long-term trends in the magnitude of variability about the mean abundance of each species are potentially of interest for conservation purposes, but will be extremely difficult to quantify in the absence of (1) stable mean densities, and (2) sampling techniques that can negate the spatial patchiness of fishes in the field.

Similarly, 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).

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. Stratification by habitat will be needed 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.

Annual monitoring should continue at least until the increase in snapper numbers observed in this 2003 survey is better understood. 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.
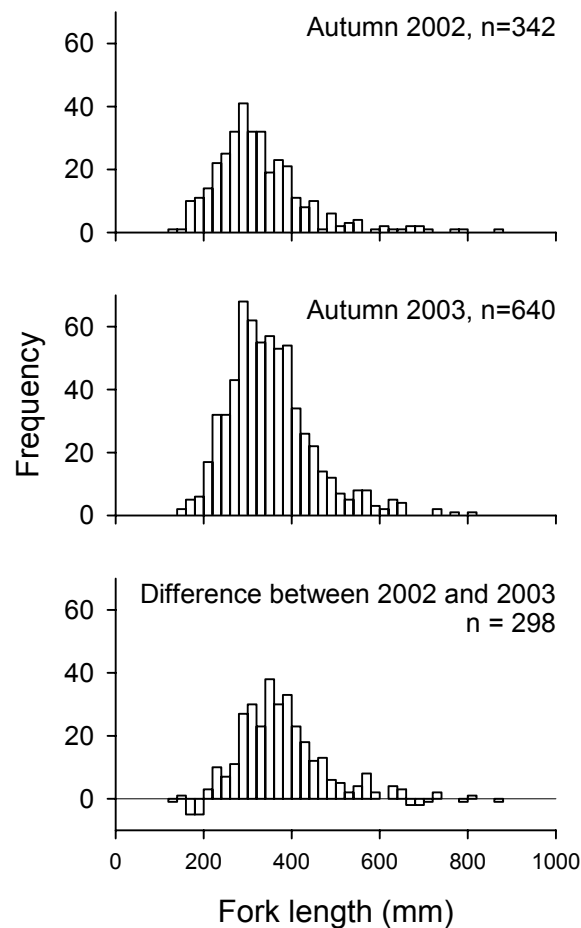

Figure 16. Change in size frequency distributions of snapper *Pagrus auratus* inside the Cape Rodney to Okakari Point Marine Reserve from autumn 2002 to autumn 2003, as measured using BUV.

Recommendations

- The fish monitoring programme should be continued with the current levels of sample replication regarded as a minimum level of effort.

- Seasonal variability is of sufficient magnitude in many of the more common species, and snapper in particular, that sampling on a six-monthly basis is desirable.

- More detailed stratification by habitat will help to determine whether large-scale habitat change inside the reserve has strong effects on the fish assemblage.

- The programme should be extended to include comparison with projected new reserves (e. g., Tiritiri Matangi) using identical sampling design and methodology. Timing of surveys must also be kept consistent between years. Comparison of this established reserve with a new reserve will provide insights as to 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, 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

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