Cape Rodney to Okakari Point Marine Reserve and Tawharanui Marine Park Reef Fish Monitoring: Autumn 2008

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Cape Rodney to Okakari Point Marine Reserve and Tawharanui Marine Park Fish Monitoring: Autumn 2008

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Executive Summary

- A survey of reef fish diversity and abundance using the UVC technique was undertaken within the Cape Rodney to Okakari Point (CROP) marine reserve and Tawharanui Marine Park (TMP) including adjacent unprotected (fished) control areas in autumn 2008. The CROP survey was a continuation of monitoring that has taken place since 2000, whereas the survey of TMP was the first formal UVC survey of this nature.

- Snapper abundance was ~ 4 times higher within CROP (2.2 per 125 m^2) than in adjacent non-reserve areas (0.5 per 125 m^2), whereas mean snapper size was around 2-fold higher within CROP (295.2 mm ± 16.3 (95 % CI)) than non-reserve areas (117.4 mm ± 13.8 (95 % CI)). Despite these differences, snapper mean abundance has declined linearly since 2003 within CROP, with 2008 numbers similar to 2002. No legal-sized snapper were observed outside of the reserve, although juveniles < 100 mm were more abundant outside of the reserve than in CROP. Blue cod abundance was also higher within CROP relative to unprotected areas, but levels have declined since 2003. This decline may be related to elevated sea surface temperatures.

- Snapper abundance did not differ significantly between TMP and non-reserve areas sampled, although the mean snapper size was ~ 2-fold higher within the reserve, i.e., 248.9 mm ± 22.8 (95 % CI) in the reserve, compared to 107.7 mm ± 9.2 (95 % CI) outside the reserve.

- The reef fish assemblage within CROP continues to be distinct from that found in adjacent fished areas, which may, in part, be related to habitat differences between the areas surveyed.

- The reef fish assemblage within TMP was not statistically distinct from fished areas, although non-reserve areas tended to display higher assemblage heterogeneity.

Key recommendations are:

Given the decline in snapper and blue cod numbers within CROP, the fish monitoring programme should be continued at one to two year intervals. The current level of sample replication is regarded as a minimum level of effort and future surveys should incorporate a balanced sampling design, as for previous surveys.

The advent of reef fish monitoring at TMP is a beneficial directive as: 1) it provides necessary baseline information should the status of TMP change from a Marine Park to a Marine Protected Area managed by DoC; 2) it provides additional information on marine reserve functioning in the outer Hauraki Gulf; and, 3) due to the consistent methodology, can be compared to CROP and other locations (Poor Knights, Hahei) that have been routinely monitored.

The effect of reserve protection on benthic communities should be assessed. Future fish surveys should be combined with a benthic monitoring program to assess changes in the invertebrate and algae communities.

Simultaneous and continued monitoring of CROP and TMP should be an important directive for DoC, as marine reserves are not static environments and given the potential expansion of coastal development in the Rodney district, the marine environment is likely to be subject to substantial pressures in the future.

Contents

1.0   Introduction

Monitoring community structure, species diversity, and the distribution and abundance of dominant species and communities through space and time is an important component of ecosystem and conservation management. Monitoring studies allow not only detection of change through space and time, but also help determine rates of change and mechanisms of change (Russ *et al*. 2005).

No-take marine reserves (Marine Protected Areas - MPAs) provide a useful tool for monitoring habitat change (Shears and Babcock 2003; Parsons *et al*. 2004) in tandem with gauging the response of marine communities and exploited species to protection (Willis *et al*. 2003; Lafferty 2004; Guidetti, *et al*. 2005). Studies in Australasia have provided convincing evidence of the conservation value of no-take MPAs through the enhancement and retention of species normally vulnerable to fishing (Babcock *et al.,* 1999; Edgar and Barrett 1999; Kelly *et al*. 2000; Willis *et al*. 2003; Shears *et al*. 2006,). However, recovery processes in marine reserves and associated trophic interactions are complex (e.g., Shears and Babcock 2003), and may vary considerably among locations. Furthermore, there is increasing awareness that monitoring studies must span sufficient spatial and temporal scales to encompass changes in oceanographic climate (Dayton *et al*. 1999; Underwood *et al.* 2000), which may strongly influence reef fish assemblages.

In recent years, the Department of Conservation (DoC) has been responsible for the collection of a large dataset for exploited species inside and outside marine reserves along the northeastern coast of the North Island (Kelly *et al*. 2000; Taylor *et al*. 2003; Usmar *et al*. 2003; Willis *et al*. 2003a; Denny & Shears 2004; Taylor *et al*. 2006; Denny 2008). These datasets provide biological data that help to evaluate the performance of marine reserves to varying levels of protection and influences the way reserves are managed. Frequent monitoring of the abundance of reef fishes within the Cape Rodney Marine Reserve (CROP), New Zealand's oldest marine reserve (gazetted in 1975) 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). Two different techniques have been traditionally used for CROP reef-fish surveys: Baited underwater video (BUV) (see Willis & Babcock 2000b, Willis *et al*. 2000) are used to survey carnivorous species (snapper and blue cod) that are difficult to survey with traditional diver mediated census techniques; and, Underwater Visual Census (UVC) transects for quantifying demersal reef species.

Results of monitoring studies at Leigh have demonstrated higher abundances of exploited species (predominantly snapper and blue cod) within protected areas relative to unprotected control areas and distinct assemblages of demersal reef species associated with reserve and non-reserve areas, thought to be related, in part, to habitat variation between these areas (Taylor *et al*. 2005).

Snapper and blue cod abundances have also been monitored within Tawharanui Marine Park (TMP) using BUV between 1997 and 2000 (Willis *et al*. 2003) and again in 2007

(unpublished data). Tawharanui Marine Park (also a no-take reserve) was established in 1992 and is situated ~ 8 km south of CROP.

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 (e.g., Shears and Babcock 2003) arising indirectly from changes in the relative density of predators (trophic cascades) (Taylor *et al*. 2005).

This report presents the results of a survey of CROP and TMP reserves and adjacent unprotected areas undertaken during autumn 2008 using UVC. The UVC technique was identical to that used in previous surveys of CROP.

Glossary of terminology

In this report the following terminology and abbreviations are used:

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).

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.

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.

2.0   Methods

2.1   Survey design

The 2008 census of the Cape Rodney to Okakari Point Marine Reserve and Tawharanui Marine Park was carried out between May 15 and June 3, 2008. A total of 16 areas were surveyed (Fig. 2.1) across the Leigh and Tawharanui coastline. For the CROP reserve and adjacent coastline these were areas 3, 4, 5, 6, 7 & 8 (within the reserve) and sites 2, 9, 10 & 12 (outside of the reserve). For the TMP and adjacent coastline these were sites 15, 16, 17 (within the reserve) and sites 18, 19, 20 (outside of the reserve).

The survey design and methods for UVC were identical to those used in past surveys (e.g., Taylor *et al.* 2005). Within each area, sampling 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.

2.2   Underwater visual census

Within each site (two per area), two divers surveyed fishes within a total of ten 25 m $\times$ 5 m transects. For each transect, 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 counted, and depending on species, sized to $\pm$ 50 mm. This methodology has been utilised for other fish surveys in New Zealand and therefore, provides users with data to compare fish assemblages from different regions. Occasionally, blue cod and spotties would follow divers between transects, and care was taken not to include these individuals in subsequent transect replicates. Depth and the percent cover occurrence of broad habitat types according to Shears *et al.* (2005) were recorded for each transect.

2.3   Data analysis

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., *n* = 10 transects were summed for each variable to obtain a single observation for each station). There were 26 fish species recorded and included in analyses of CROP and TMP and associated control areas. For CROP there was a total of 20 multivariate observations, consisting of 2 stations within each of 10 areas, with 6 areas located inside the reserve (areas 3-8) and 4 areas located outside the reserve (areas 2, 9, 10 and 12). However, for analyses that required a balanced dataset, reserve sites 3 and 8 were not included in the analysis. For TMP there was a total of 12 multivariate observations, consisting of 2 stations within each of 6 areas, with 3 areas located inside the reserve (areas 15-17) and 3 areas located outside the reserve (areas 18-20).

All multivariate statistical tests were based on Bray-Curtis dissimilarities (Bray and Curtis 1957) calculated among observations for data transformed to $y' = \ln(y+1)$. Relative dissimilarities in the fish assemblages observed at different stations were visualized using principal coordinate analysis (PCO, Gower 1966). 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 (9999 permutations) for each individual term in the model (Anderson 2001b). Data were also examined for homogeneity of multivariate dispersions using the computer programme PERMDISP (Anderson 2004).

The effect of marine reserve status on reef fish assemblages were also tested using canonical analysis of principal coordinates (CAP, Anderson and Willis 2003, Anderson and Robinson 2003). CAP is a constrained ordination procedure, which finds an axis through the multivariate cloud that is best a discriminating group differences in multivariate space, if differences do exist. CAP is effectively a PCO followed by a traditional canonical discriminant analysis on a number of PCO axes. Correlations of individual species with the canonical axis corresponding to "Status" was used as an indication of the species responsible for the differences in species assemblage patterns between reserve and non-reserve sampling sites.

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 SAS statistical software (SAS 1999).
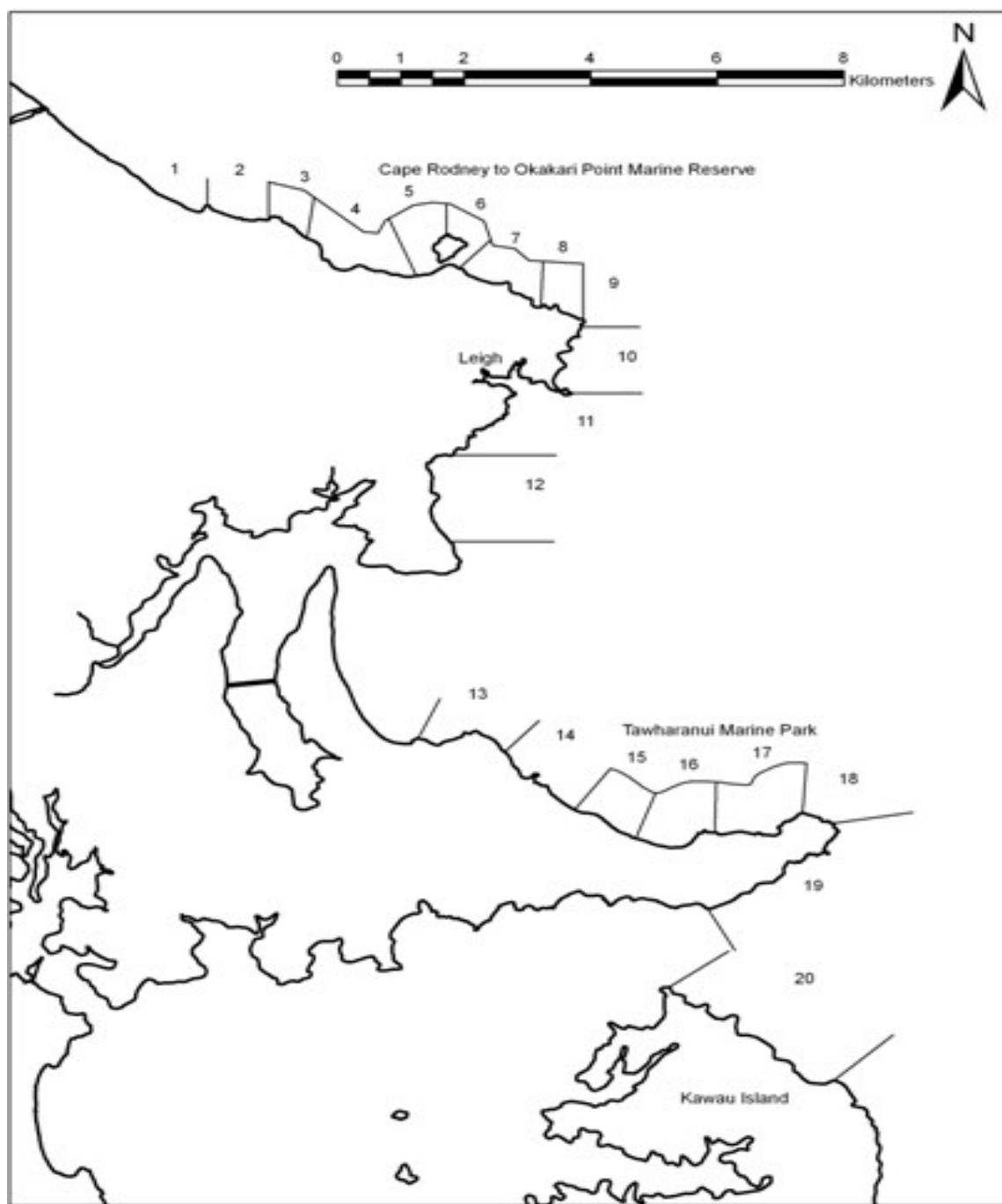
Figure 2.1. Map of survey areas across the Leigh and Tawharanui regions surveyed in 2008. CROP reserve areas surveyed were 3 to 8, with non-reserve areas 2, 9, 10, and 12. TMP areas surveyed were areas 15 to17, with non-reserve areas 18 to 20.

3.0   Results

3.1   Cape Rodney to Okakari Point marine reserve and control areas: Community-level patterns

In 2008, the reef fish assemblage structure was statistically different[1] (PERMANOVA – Anderson 2005) between reserve and non-reserve locations (Status, $F = 4.89$, $P < 0.05$), as were assemblages among areas within reserve and non-reserve areas surveyed (Area (Status), $F = 2.49$, $P < 0.001$). The difference between reserve and non-reserve areas, as for previous surveys, may be related to the dispersion of the fish assemblages. Results from dispersion analysis (PERMDIS) (Anderson 2004) indicated a statistically significant difference between reserve and non-reserve areas ($F = 2.36$, $P < 0.05$) and the PCO ordination (Fig. 3.1) suggests that non-reserve sites have a much broader spread than reserve sites, which with the exception of Site 5 (North Reef), appear broadly similar to each other (Fig. 3.1). Within the reserve, average dissimilarity among areas was 29 % compared to 38 % for non-reserve.

Canonical analysis indicated a significant overall effect of reserve status on the fish assemblages (Fig. 3.2 canonical correlation, $\delta^2 = 0.65$, $P < 0.05$). A range of species had higher frequencies and were accordingly more-representative of fish assemblages within the reserve including parore, snapper, butterfish, blue cod, banded wrasse and sweep (Table 3.1).   Assemblages outside of the reserve had higher frequencies of spotty, hiwihiwi, demoiselles, trevally, jack mackerel, goatfish, and leatherjacket (Table 3.1).

As suggested for previous surveys (Taylor *et al*. 2005), larger assemblage variability outside the reserve may be related to the prevalence of different habitat types and potentially greater habitat variability, compared to inside. Based on percent occurrence, the kelp *Ecklonia radiata* (hereafter *Ecklonia*) was by far the most widespread habitat type (across areas surveyed within CROP followed by mixed algae (both *Ecklonia* and mixed algae accounting for over 90 % of habitats recorded), urchin barrens, turfing algae, and sand (all < 5 %) (Fig. 3.3). Conversely, urchin grazed barrens was the dominant habitat type (> 60 %) in non-reserve areas followed by *Ecklonia* forest, mixed algal habitat, cobbles, sand, and turf (Fig. 3.3).

Higher variability in fish assemblages in areas outside the reserve could also be due to patchiness in the occurrence of species. On average there were significantly fewer species observed in areas outside the reserve (mean = 13.7 ± 0.6 SE), compared to areas inside the reserve (mean = 16.3 ± 1.1 SE) ($F_{1,8} = 6.37$, $P = 0.027$). The mean total abundance of fish recorded by UVC was greater inside (mean = 687.7 ± 89.7 SE) compared to outside the reserve (mean = 644.4 ± 84.36 SE), but this difference was not statistically significant ($F_{1,8} = 0.92$, $P = 0.374$).

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[1] Statistical analysis was undertaken using reserve sites 4-7 and non-reserve sites 2,9,10 and 12, for PERMMANOVA and PERMDISP, thereby creating a balanced dataset. Data from all sites were used for PCO and CAP analysis.
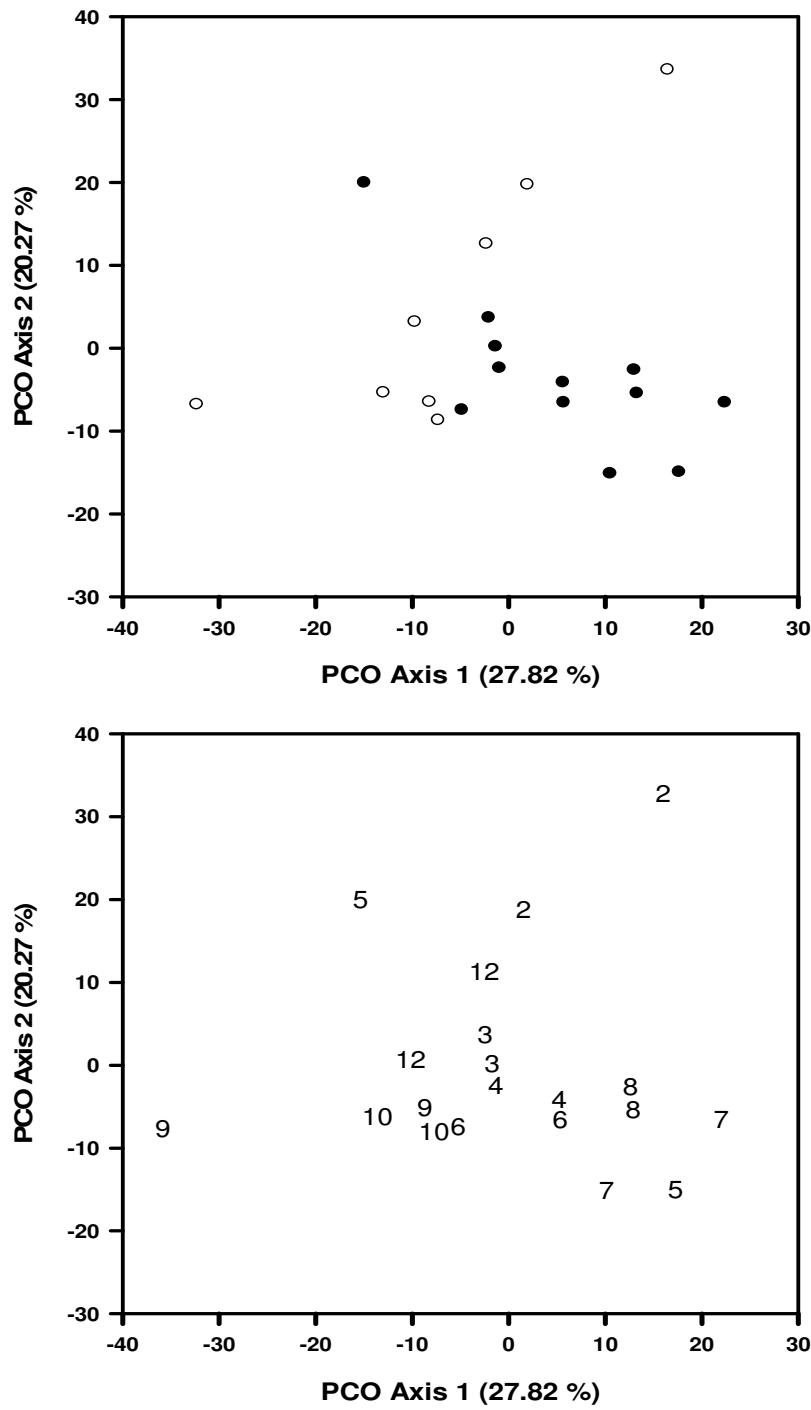
Figure. 3.1. Ordination plot of the first two PCO axes (explaining 48.1 % of the original variability) based on Bray-Curtis dissimilarities of ln(y+1) transformed species abundance data (26 species), showing assemblages at different stations with labels for (a) CROP reserve (dark symbols) versus non-reserve (open symbols) status or (b) reserve areas 3 through 8 (with 2 stations per area) and non-reserve areas 3, 8,9,10 and 12.
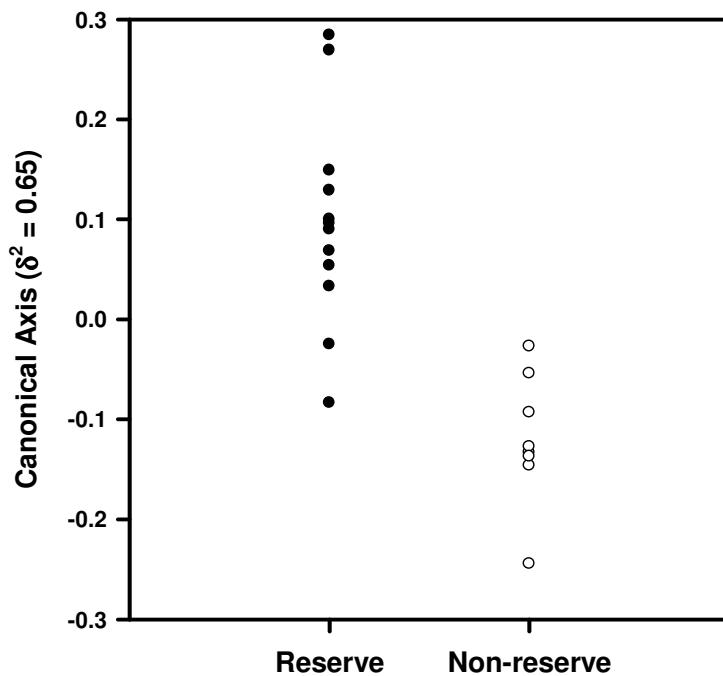
Figure 3.2. Plot of the canonical axis from a CAP constrained ordination to discriminate fish assemblages from CROP reserve versus non-reserve stations. The discriminant analysis was done on the first $m = 8$ PCO axes (which explained 99.99% of the original variability) from Bray-Curtis dissimilarities of ln(y+1) transformed species abundances (26 species).

Table 3.1. 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
Parore	*Girella tricuspidata*	0.635
Snapper	*Pagrus auratus*	0.533
Butterfish	*Odax pullus*	0.503
Banded wrasse	*Notolabrus fucicola*	0.284
Blue cod	*Parapercis colias*	0.279
Negative correlation (non-reserve)		
Demoiselles	*Chromis dispilus*	-0.678
Trevally	*Pseudocaranx dentex*	-0.435
Jack Mackerel	*Trachurus novaezelandiae*	-0.409
Goatfish	*Upeneichthys lineatus*	-0.345
Leatherjacket	*Parika scaber*	-0.277
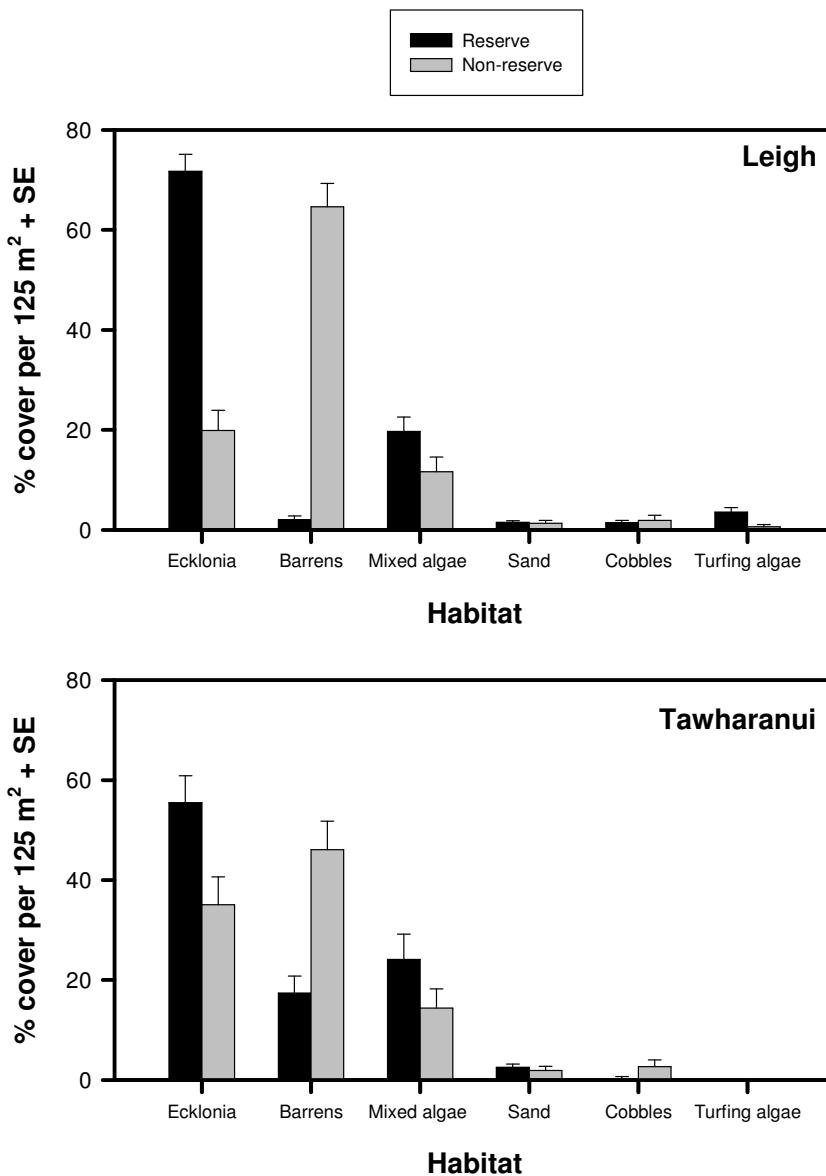
Figure 3.3. Main habitat types (pooled across areas) for Leigh and Tawharanui reserve and non-reserve locations surveyed in 2008.

3.2 Tawharanui Marine Park and control areas: Community-level patterns

Reef fish assemblages were statistically significant (based on PERMANOVA analysis) among areas surveyed within the Tawharanui Marine Park and among unprotected areas (Area (Status), $F = 3.45$, $P < 0.001$), however reef fish assemblages were not significantly different in relation to Status ($F = 1.20$, $P = 0.220$). As for the Leigh coastline surveyed, non-reserve fish assemblages were more-variable than the reserve assemblages surveyed, e.g., areas 18 and 19 appearing dissimilar to the other non-reserve

site (Site 20) and reserve sites surveyed, which are clustered to the centre of the ordination (Fig. 3.4).

Dispersion analysis (PERMDIS – Anderson 2004) indicated statistically significant differences between reserve and non-reserve areas ($F = 2.36$, P < 0.05) with average dissimilarity among areas in the reserve being 37 % compared to 55 % for non-reserve areas.

Canonical analysis indicated no statistically significant overall effect of reserve status on the fish assemblages (Fig. 3.5, canonical correlation, $\delta^2 = 0.680$, $P = 0.425$). However, butterfish, spotty, snapper, parore, and blue maomao had higher frequencies and were accordingly more-representative of fish assemblages within the reserve (Table 3.2), whereas assemblages outside of the reserve had higher frequencies of eagle ray, bigeye, demoiselles, and banded wrasse (Table 3.2).

Species diversity was slightly higher in areas outside the reserve (mean = 14.3 ± 2.3 SE), relative to areas inside the reserve (mean = 13.3 ± 0.3 SE), but this difference was not statistically significant ($F_{1,4} = 1.2$, $P = 0.85$). Similarly, the mean total abundance (excluding pelagic species) was also slightly higher for non-reserve areas (mean = 479 ± 61.3 SE) compared to the reserve (445.33 ± 67.9 SE); again this difference was not statistically significant ($F_{1,4} = 0.92$, $P = 0.85$).

Habitats with the highest mean percent cover within TMP averaged across the areas surveyed were *Ecklonia* forest, mixed algal habitat and urchin barrens, with sand and cobbles comprising < 3 % (Fig. 3.3). In non-reserve areas, barrens habitat had the highest mean percent cover followed by *Ecklonia* forest and mixed algal habitat. Cobbles and sand comprised < 3 % (Fig. 3.3). Differences between barrens and *Ecklonia* habitat between TMP and non-reserve areas was not as large as for CROP and turfing habitat was not recorded along the Tawharanui and Kawau Island coastline.
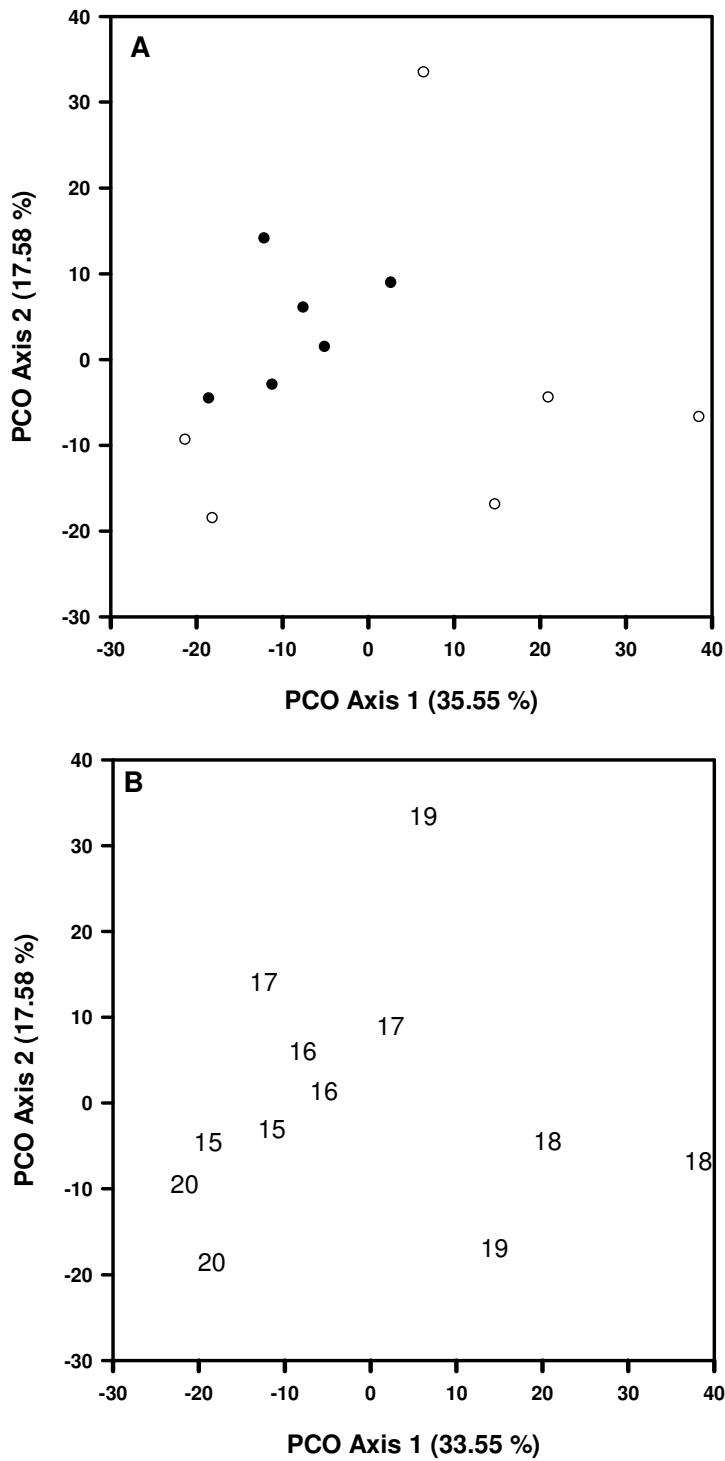
Figure 3.4. Ordination plot of the first two PCO axes (explaining 51.13% of the original variability) based on Bray-Curtis dissimilarities of ln(y+1) transformed species abundance data (26 species), showing assemblages at different stations with labels for (a) TMP (dark symbols) versus non-reserve (open symbols) status or (b) TMP reserve areas 15 through 17 (with 2 stations per area) and non-reserve areas 18 through 20.
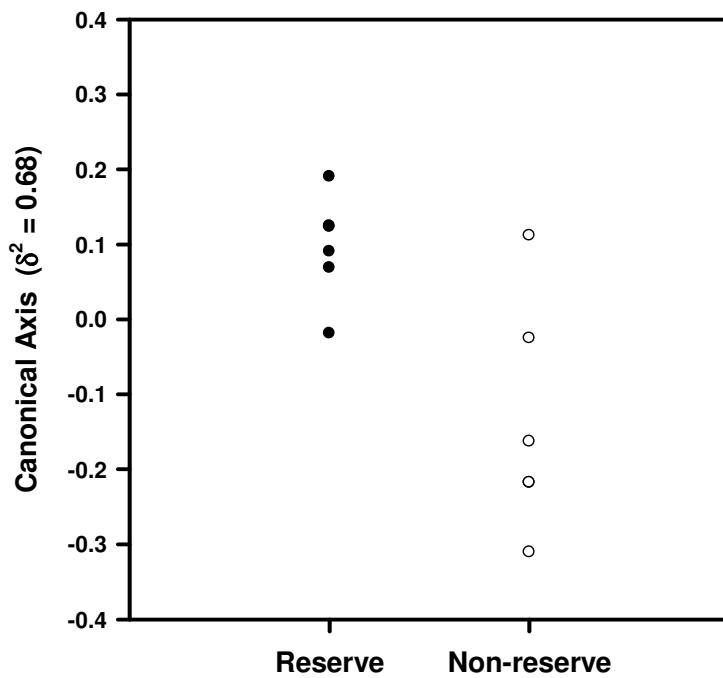
Figure 3.5. Plot of the canonical axis from a CAP constrained ordination to discriminate fish assemblages from TMP reserve versus non-reserve stations. The discriminant analysis was done on the first $m = 9$ PCO axes (which explained 99.99% of the original variability) from Bray-Curtis dissimilarities of $\ln(y+1)$ transformed species abundances (26 species).

Table 3.2. 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
Butterfish	*Odax pullus*	0.503
Snapper	*Pagrus auratus*	0.477
Parore	*Girella tricuspidata*	0.393
Blue maomao	*Scorpis violaceus*	0.273
Sweep	*Scorpis lineolatus*	0.210
Negative correlation (non-reserve)		
Eagle ray	*Myliobatus tenuicaudatus*	-0.612
Big eye	*Pempheris adspersus*	-0.585
Demoiselles	*Chromis dispilus*	-0.320
Banded Wrasse	*Notolabrus fucicola*	-0.262

3.3 Cape Rodney to Okakari Point marine reserve and control areas: Individual species

Comparisons of individual taxa between the CROP reserve and non-reserve was variable between areas surveyed in 2008, which match patterns through time (Fig's 3.6-3.11). In 2008, snapper were ~ 4-times higher within the reserve (2.2 per 125 m^2) than non-reserve (0.5 per 125 m^2) and on average ~ 2–fold larger (295.2 mm ± 16.3 (95 % CI) reserve) compared to (117.4 mm ± 13.8 (95 % CI) non-reserve). Differences in abundance and size were statistically significant ($P < 0.001$; paired Wilcoxon signed rank test). The difference in size among reserve and non-reserve areas is reflected further in the size frequency distributions (Fig. 3.12) with the reserve population ranging from 100 mm to 600 mm, whereas no legal-sized snapper occurred within sample transects outside of the reserve. Conversely, juveniles < 100 mm had a higher frequency in non-reserve areas sampled along the Leigh coastline and were common on patch reef adjacent sandy substratum.

Despite a higher abundance and frequency of larger snapper in CROP compared to non-reserve areas in 2008, there has been a steady (and linear) decline in abundance within the reserve since 2003; although levels remain higher than that recorded in autumn 2002 when snapper abundance in the reserve was ~ 1 per 125 m^2 (Fig. 3.6). Snapper numbers outside of the reserve have also declined linearly since 2003, although 2008 levels are based on four areas being surveyed, whereas previous years are based on six areas being surveyed.

Similarly, blue cod have also declined across the reserve areas sampled since 2003, with 2008 levels similar to that recorded in both 2001 and 2002 autumn surveys, i.e., < 0.3 per 125 m^2. Increased sea surface temperate (Fig. 3.13) over these periods has been touted as playing a causative role in the decline of blue cod numbers across the Leigh area.

Other species that have demonstrated a decline in abundance since the previous two surveys (2003 and 2005) include red moki (reserve and non-reserve), spotty (reserve and non-reserve areas), silver drummer (reserve), whereas banded wrasse, parore, and blue maomao have all increased in abundance within the reserve, but have declined in non-reserve areas (Fig's 3.4-3.8). Levels of butterfish have essentially remained unchanged for both reserve and non-reserve areas (Fig.3.11). Any declines in abundance for taxa outside the reserve in 2008 relative to previous years, however, should be viewed with caution, due to a lower level of replication used in the 2008 survey.

Abundance patterns of taxa such as kahawai, sweep, and jack mackerel, which have been highly variable among reserve and non-reserve areas from the beginning of the monitoring programme were also highly variable in 2008 (Fig's 3.8, 3.10). This is not unexpected as these taxa are commonly found in large schools and are highly mobile.

### 3.4   Tawharanui Marine Park and control areas: Individual species

Mean snapper abundance in 2008 was slightly higher within TMP (0.8 ± 0.2 SE per 125 m^2) compared to outside (0.6 ± 0.5 SE per 125 m^2) (Fig. 3.6), but this difference was not statistically significant (P = 0.343; paired Wilcoxon signed rank test). Snapper mean size was 2-fold higher for the reserve snapper sample population (248.9 mm ± 22.8 (95 % CI) compared to the non-reserve sample population (107.7 mm ± 9.2 (95 % CI) and this difference was statistically significant (P < 0.001; paired Wilcoxon signed rank test). This difference is depicted further in the size frequency data (Fig. 3.12), with the reserve population having a higher frequency of larger snapper than non-reserve. However, during the survey, larger snapper were often observed outside of transects and commonly displayed diver-negative responses. As was observed for non-reserve areas adjacent to CROP, there was a higher frequency of juveniles < 100 mm, which were also common on patch reef adjacent to sandy substratum habitat.

Other taxa with higher abundances in reserve areas compared to non-reserve included red moki, banded wrasse, spotty, parore, blue maomao and butterfish (Fig's 3.7-3.11). Differences in abundance were only statistically significant for spotty and banded wrasse; P < 0.05 respectively (paired Wilcoxon signed rank test).
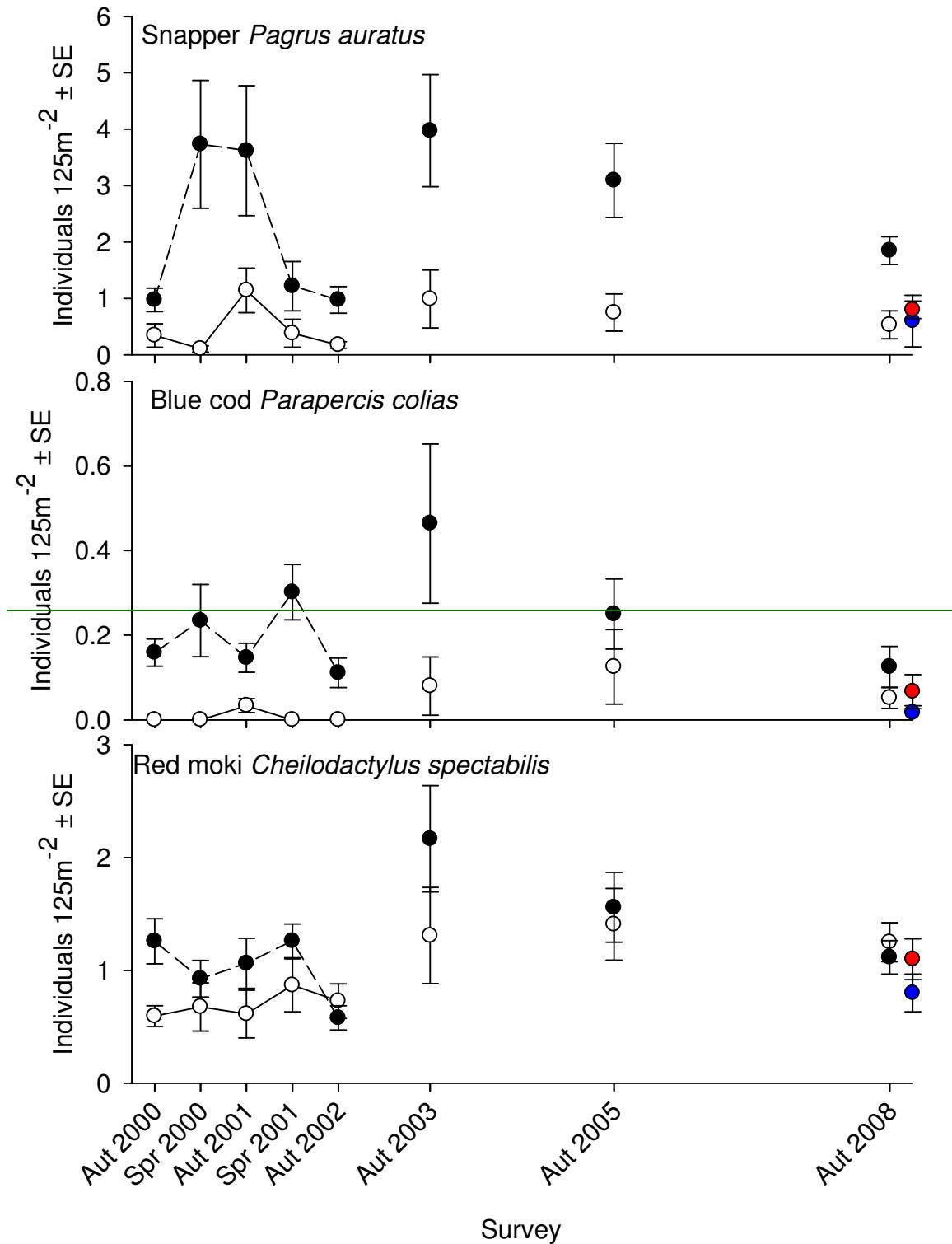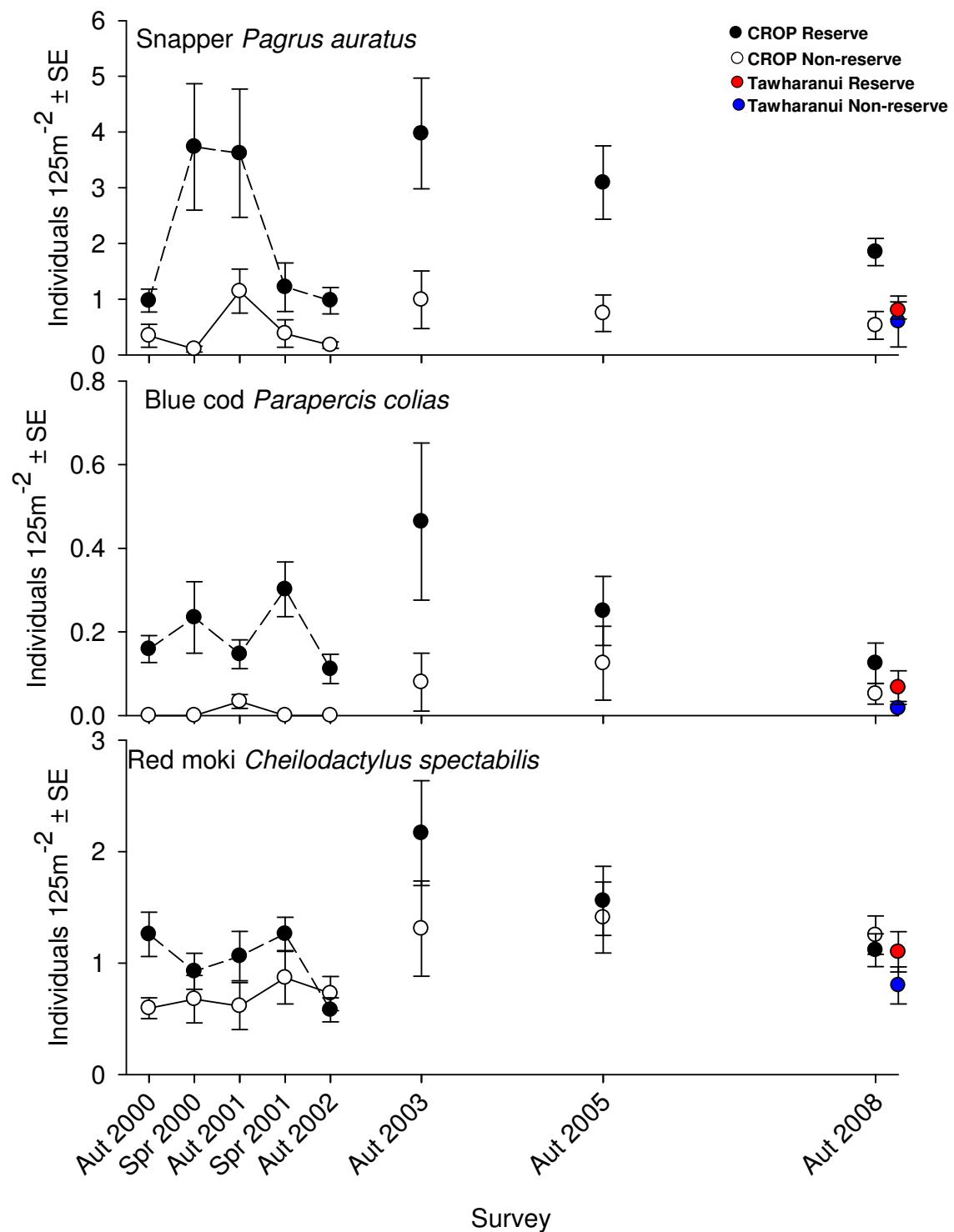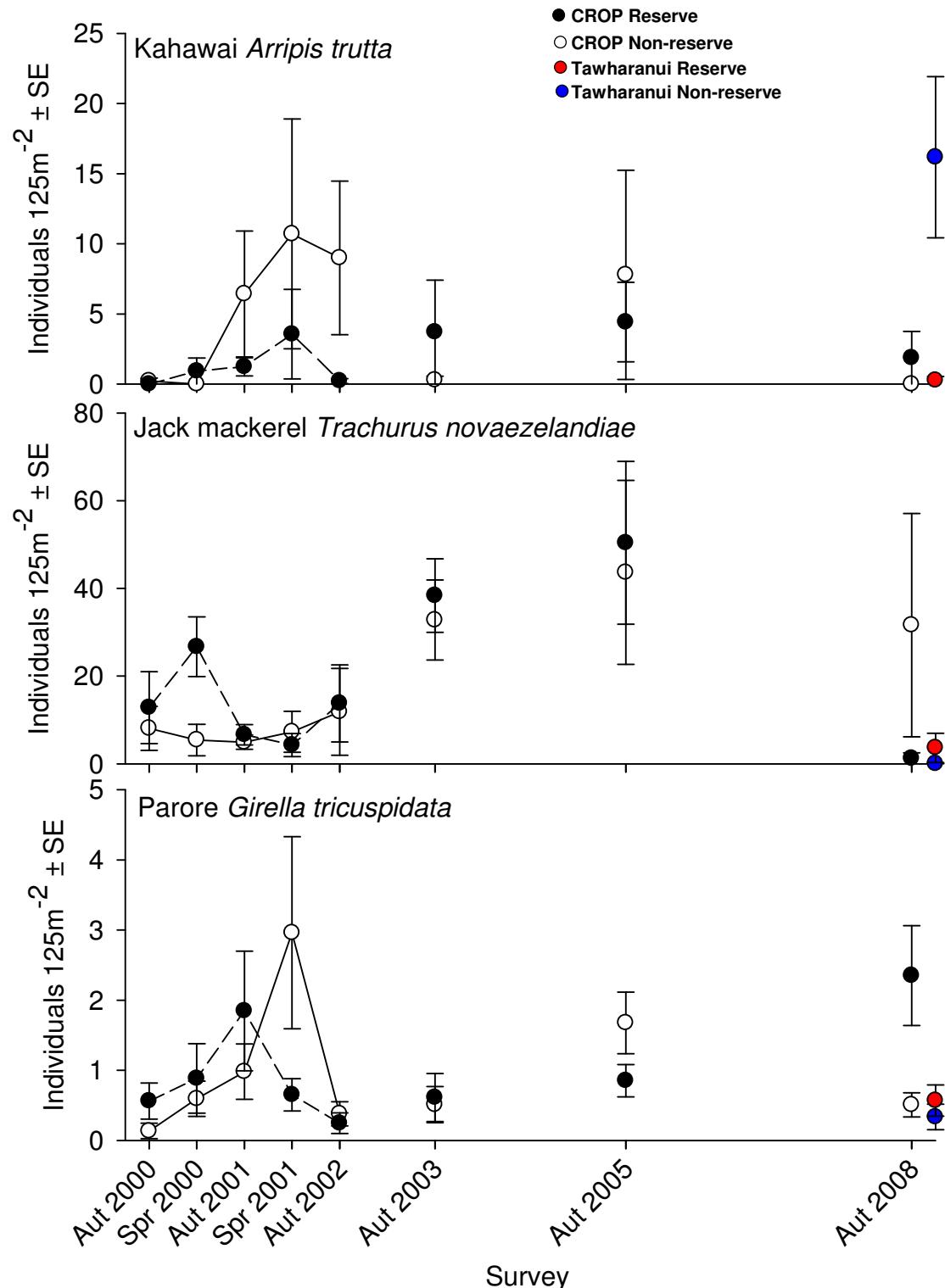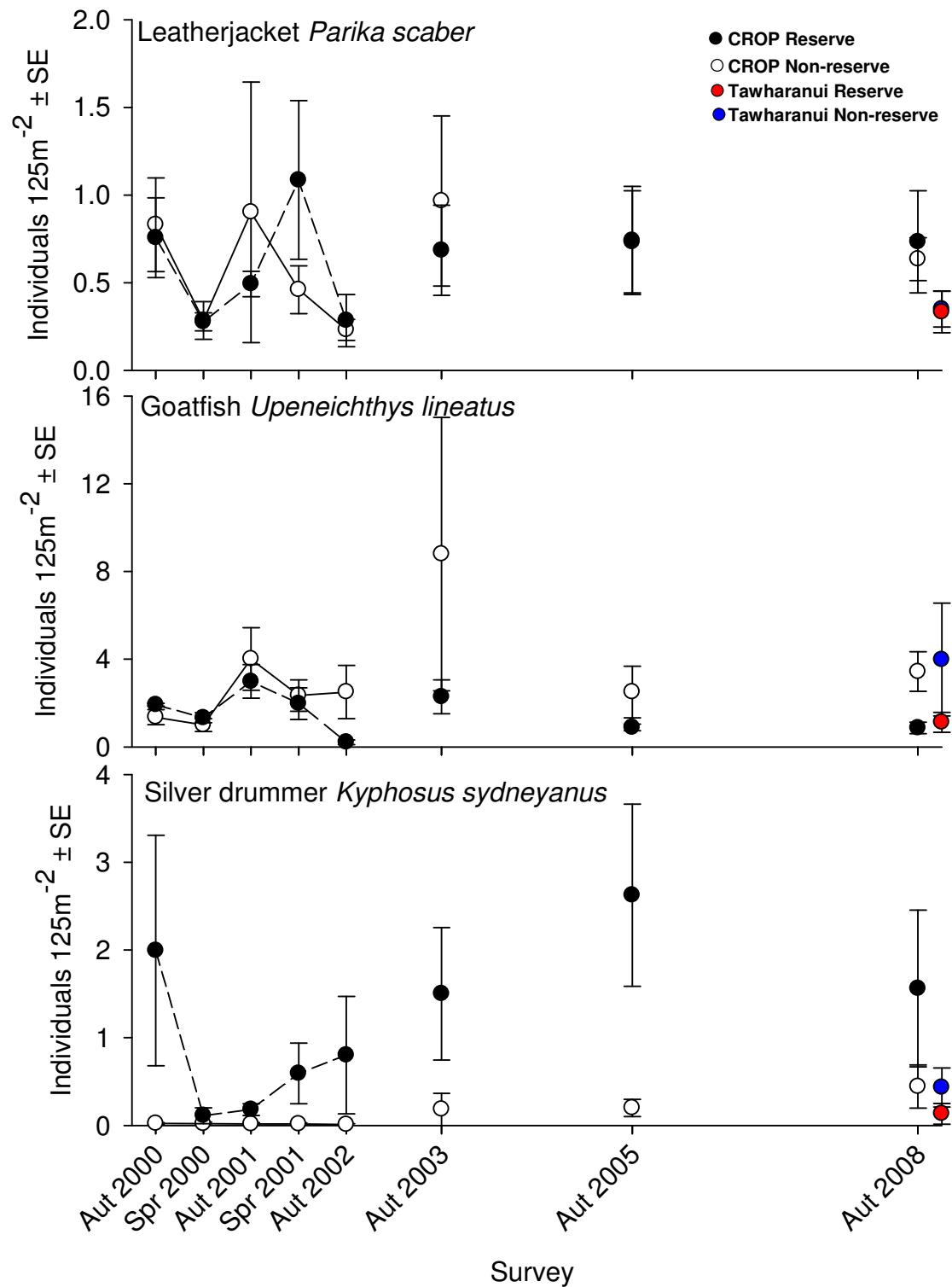
Figure 3.6. Long term trends in the densities of snapper, blue cod, and red moki inside and outside the Cape Rodney to Okakari Point Marine Reserve, as measured using UVC between 2000 and 2008 and inside and outside Tawharanui Marine Park in 2008. Note: *y* axis scale differs among plots.
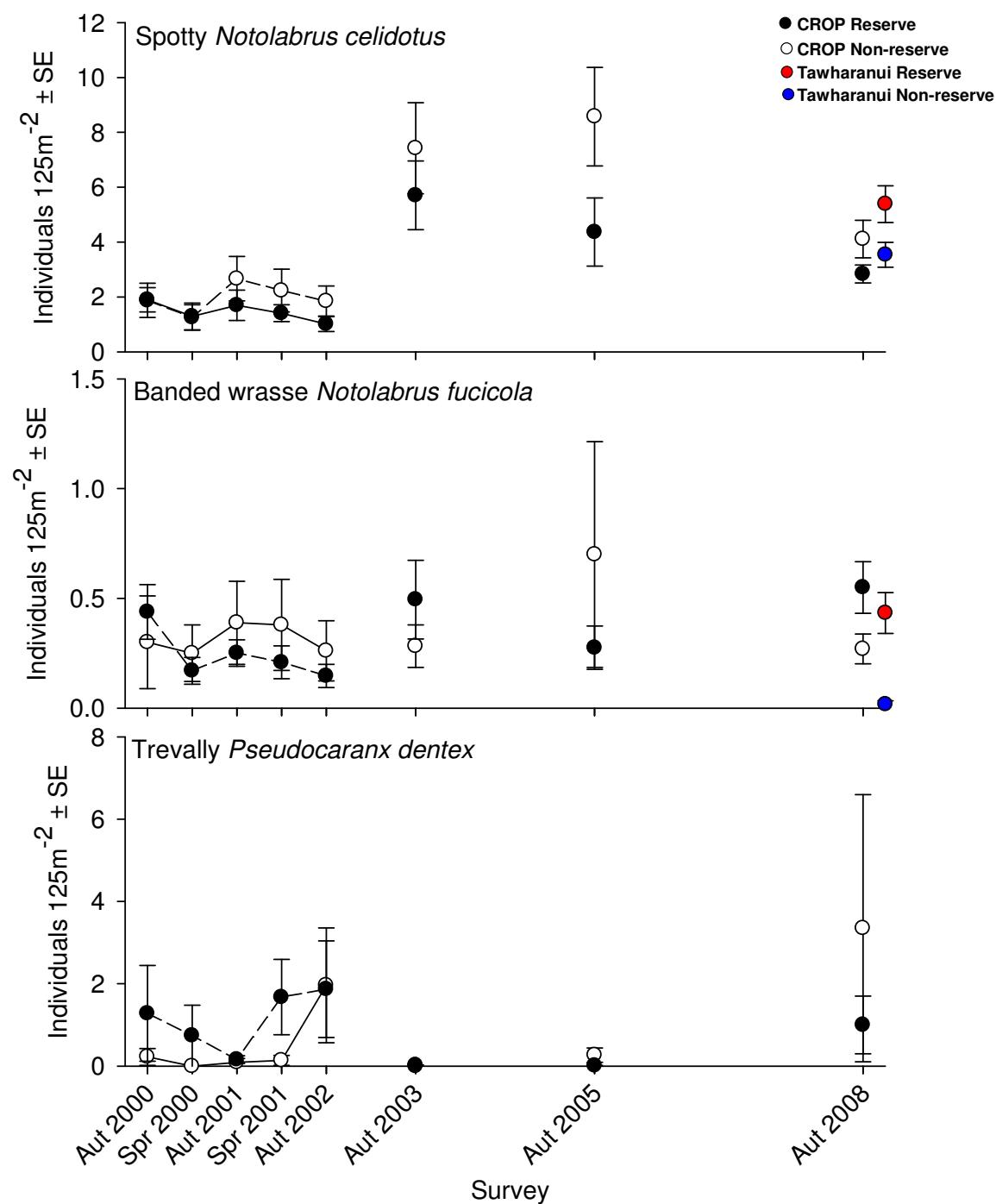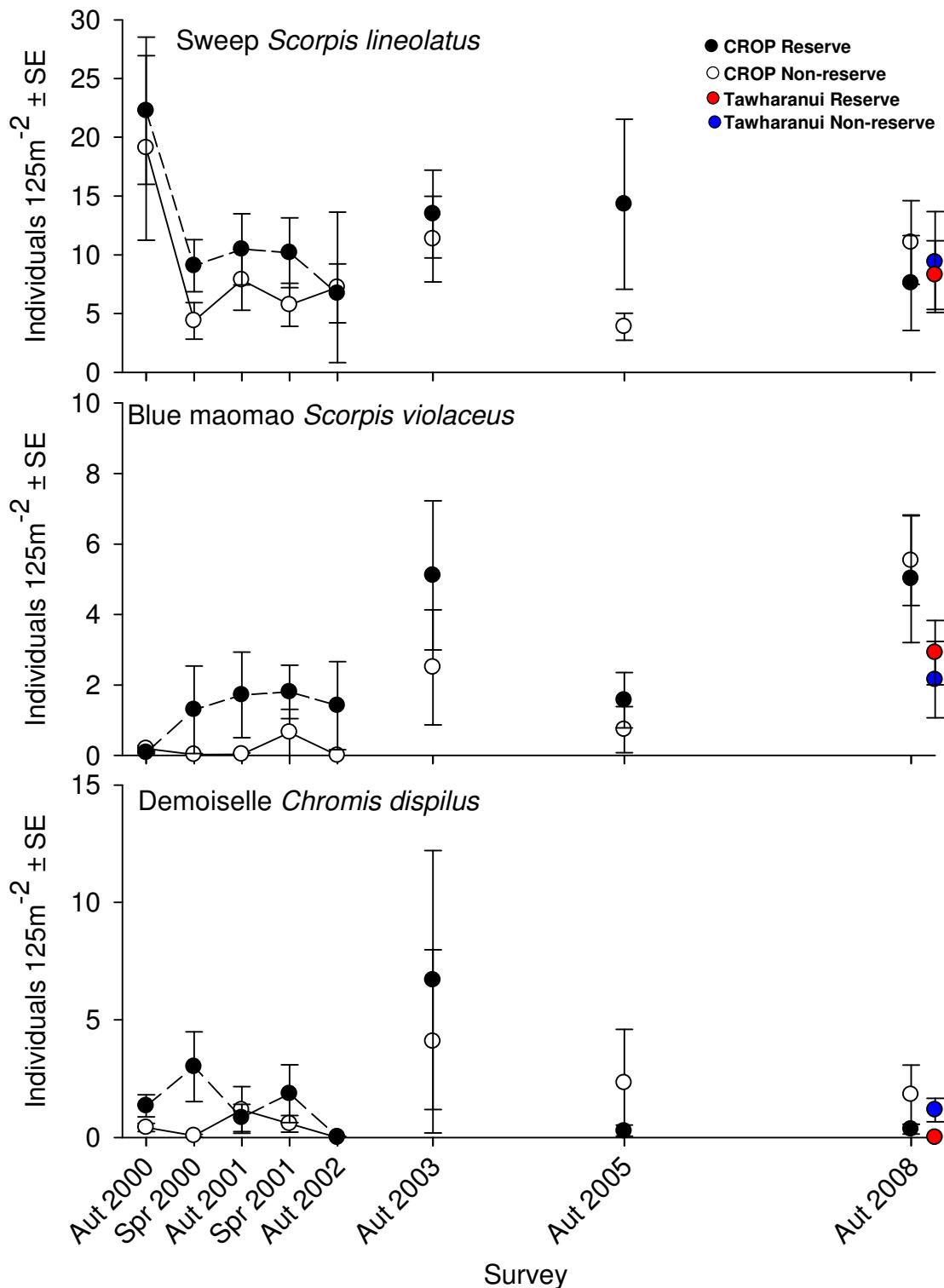
Figure 3.7. Long term trends in the densities of spotty, banded wrasse, and trevally inside and outside the Cape Rodney to Okakari Point Marine Reserve, as measured using UVC between 2000 and 2008 and inside and outside Tawharanui Marine Park in 2008. Note: *y* axis scale differs among plots.

Figure 3.8. 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 between 2000 and 2008 and inside and outside Tawharanui Marine Park in 2008. Note: *y* axis scale differs among plots.

Figure 3.9. 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 between 2000 and 2008 and inside and outside Tawharanui Marine Park in 2008. Note: *y* axis scale differs among plots.

Figure 3.10. 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 between 2000 and 2008 and inside and outside Tawharanui Marine Park in 2008. Note: *y* axis scale differs among plots.
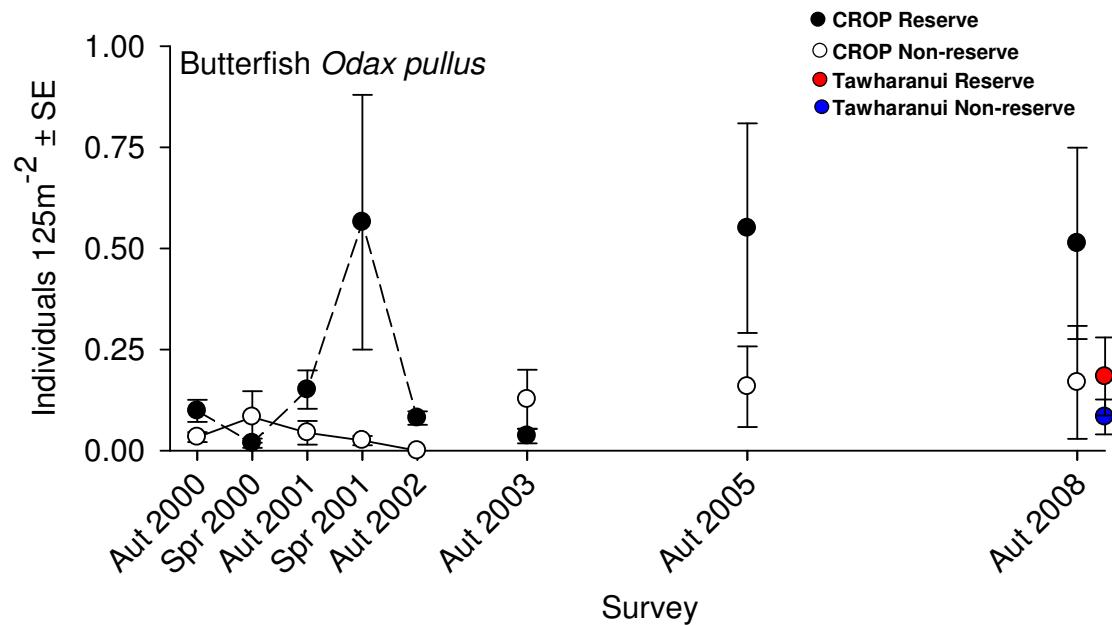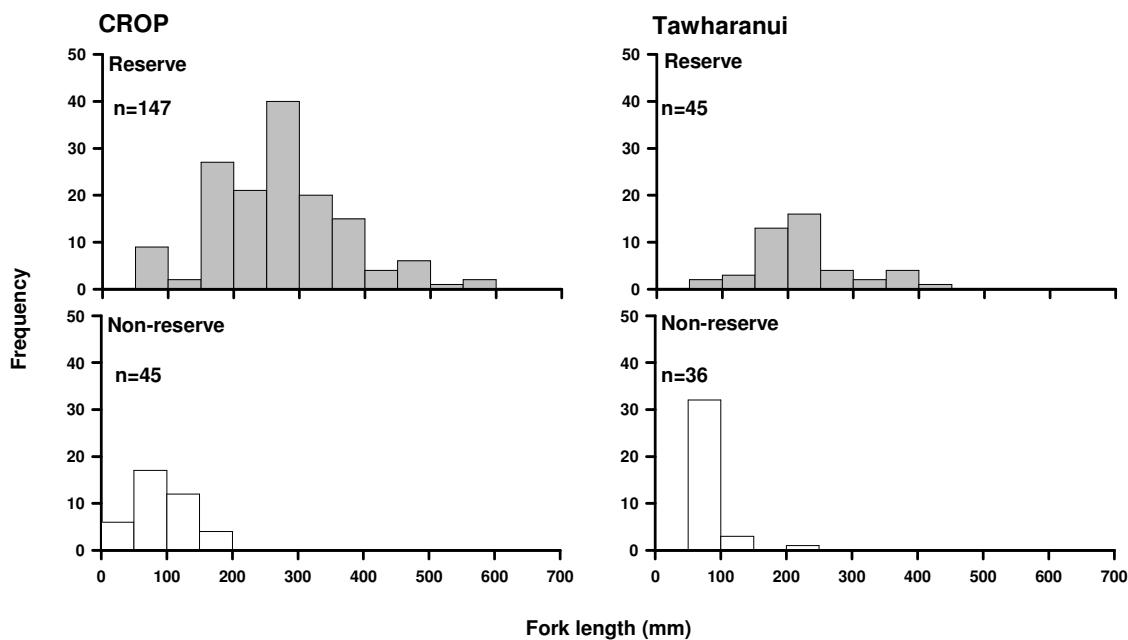
Figure 3.11. Long term trends in the density of butterfish inside and outside the Cape Rodney to Okakari Point Marine Reserve, as measured using UVC between 2000 and 2008 and inside and outside Tawharanui Marine Park in 2008.

Figure 3.12. Size frequency of snapper inside and outside the Cape Rodney to Okakari Point Marine Reserve, and inside and outside Tawharanui Marine Park in 2008, as measured using UVC.
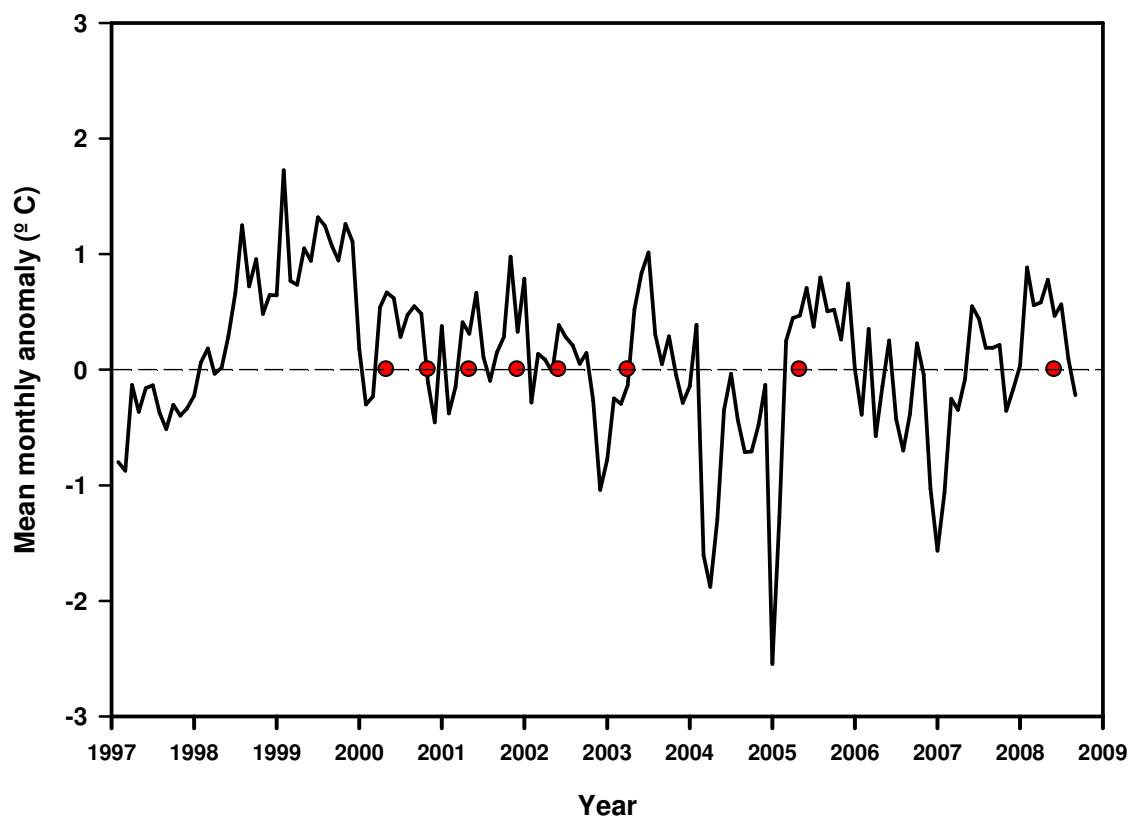
Figure 3.13. Sea surface temperature anomalies from Leigh (based on long term average 1967-97). Red dots denote UVC surveys that have been undertaken in CROP.

4.0 Discussion

The survey of reef fish abundance and diversity within the Cape Rodney to Okakari Point (CROP) reserve in autumn 2008 was the eighth survey across this area of coastline since 2000, and the most recent survey since 2005. The survey of Tawharanui Marine Park (TMP) and associated control areas was the first formal UVC survey for this area of coastline. A BUV survey was carried out in 2007, and is presently being analysed (DoC unpublished data). Other MPAs that are routinely surveyed for reef fish abundance in north-eastern New Zealand include Hahei (Taylor *et al*. 2006) and the Poor Knights Island (Denny *et al*. 2004). These surveys often use a combination of BUV and UVC techniques to quantify reef fish diversity and abundance and to estimate the size of heavily fished species (snapper and blue cod).

Mirroring patterns emanating from previous surveys, reef fish abundance was highly variable across all areas surveyed, irrespective of "Status". However, fish assemblages within CROP and TMP appeared different to, and were less variable, than non-reserve assemblages. In 2008, the CROP reserve had significantly higher abundances of snapper and blue cod (species that are actively fished along the coastline) compared to non-reserve areas. Snapper mean size was also significantly higher across within the reserve, with no legal-sized snapper (> 270 mm FL) recorded in non-reserve areas. Snapper mean size was also higher within TMP, and similarly no legal-sized snapper were recorded in non-reserve areas, although mean abundances were not different between the reserve and unprotected control areas.

Temporal data from CROP reserve surveys indicate a linear decline in the mean abundance of snapper and blue cod has taken place over the last five years, i.e., between 2003 and 2008, however present levels do fall within the long-term range for both of these species. Researchers have suggested that while snapper site fidelity is high within CROP (Parsons *et al*. 2003) snapper may move beyond the reserve boundary at certain times of the year associated with spawning (Willis *et al*. 2003), thus a certain proportion of this population is likely to be susceptible to fishing. Willis *et al*. (2003) further suggest that the observed recovery of snapper populations within MPAs is largely the result of immigration of individuals from fished areas that take up residency within reserves, rather than juvenile recruitment *per se*.

The higher abundance of juvenile snapper in non-reserve locations (Leigh and Tawharanui) within patch reef surrounded by sand indicates that these areas of coastline may contain more-favorable habitat and food for this life history stage. Thrush *et al*. (2002) suggests that small-scale habitat structure (depressions, burrows, shells, boulders, cobbles, and sand waves) within soft-sediment habitats positively influences the abundance of juvenile snapper through provision of feeding areas and refuge from predation.

In past surveys, blue cod numbers have been suggested to be negatively correlated with higher seas surface temperatures, i.e., numbers declined between 1997 and 1999 and 2003 and 2005 when sea surface temperatures were somewhat higher than average and

remained stable between 2000 and 2002 when temperature was more constant (Taylor *et al*. 2005).   Higher than average sea surface temperatures between 2007 and 2008 and low blue cod frequencies in 2008 match this general pattern, although we are sympathetic to the views of Taylor *et al.* (2005), in so far as a much longer data time-series would be necessary to test the hypothesis

The role of habitat type in influencing reserve and non-reserve fish assemblages has been proposed to be important in previous surveys of reserve and non-reserve areas (Taylor *et al*. 2005) and has been specifically quantified is purposely designed studies (see Anderson and Millar 2004, Williams *et al*. 2008).   Given that the reserve reef sites surveyed, particularly in CROP, are almost exclusively dominated by macroalgae (primarily *Ecklonia*, which is dominant at depths > 4 m Shears and Babcock 2003), it is not surprising that the differences in fish assemblage structure occurs between reserve and non-reserve areas. Presently, it is not clear what the effect of reduced urchin barrens within the reserve (associated with higher predation rates of urchins by snapper and spiny lobster (Shears and Babcock 2002)) will have on reef-fish assemblages and snapper numbers over the longer term. However, habitat changes due to the effects of increased densities of snapper have been tentatively implicated in the decline of blue cod at CROP (Willis *et al*. 2003).

Species with higher abundances or frequencies commonly associated with barrens habitat include spotty (*Notolabrus celidotus*), banded wrasse *(Notolabrus fucicola),* parore (*Girella tricuspidata),* and silver drummer (*Kyphosus sydneyanus*), with leatherjacket (*Parika* scaber)*, demoiselle (*Chromis dispilus),* jack mackerel (*Trachurus novaezelandiae),* porae (*Nemadactylus douglasii*), and butterfish (*Odax pullus*) (having higher abundances within kelp forests (Anderson and Millar 2004).   Considering the above findings, it is interesting to note (as suggested in previous CROP survey reports) that silver drummer*,* parore, and spotty have higher abundances in CROP relative to non-reserve areas.   One explanation for the continually high levels of silver drummer and parore may be related to an increase in the abundance of food, given that macroalgal habitat has increased in spatial extent within the reserve.   Reef fish diversity was also higher within CROP in 2008, a trend consistent with other surveys.   A recent study within CROP demonstrated that kelp habitat was associated with higher reef fish diversity (Williams *et al*. 2008). Barrens habitat was however not surveyed in that study, which is a comparison that would be necessary in the context of this present study (also see Anderson and Millar 2004).

Main habitat types were also appreciably different between areas inside and outside of TMP, but these dissimilarities were not as large as for CROP. Main habitats outside the reserve were a combination of mixed algae, *Ecklonia* and urchin barrens habitat, whereas those inside the reserve were predominately *Ecklonia* forest or mixed algae. Given the difference in habitat types outside of the reserve one would expect that fish assemblages would differ between reserve and non-reserve areas, although despite higher variability among non-reserve sites, analysis indicated no significant difference between reserve and non-reserve assemblages.

Because fish abundance is influenced by multiple factors such as climate, habitat type and life history traits (migration, ontogeneic shifts, predation, etc), explanations for patterns are not easily discernable, particularly as there is negligible temporal information for TMP and associated non-reserve locations. For TMP, a degree of underestimation may be evident with regard to snapper numbers, as adult snapper were commonly observed outside of the sample transects and displayed diver-negative responses. Results of BUV data may provide a more robust estimate of snapper numbers within the TMP relative to unprotected controls as they come to hand.

For the present survey, the UVC technique was useful for making broad comparisons among areas and detecting changes in fish assemblages. Because of various biases associated with the technique, Taylor *et al*. (2005) suggest these type of surveys can engender as many questions as they provide answers, as different species occupy different habitats, have different modes of behaviour (e. g., solitary versus schooling), and respond to divers in different ways. While these problems are very real, UVC surveys still remain a cheap and effective tool to quantify reef fish abundance and diversity within MPAs and corresponding control areas.

The decline in snapper and blue cod densities within CROP and the present lack of temporal data for TMP highlights the need for consistent and regular monitoring of fish assemblages in both of these MPAs. The incorporation of TMP into the long-term reef fish monitoring will be invaluable for placing trends in abundance and diversity into a larger geographical framework. For future CROP surveys the inclusion of all sites (i.e., the two non-reserve sites not surveyed in 2008) should be mandatory and will also ensure a balanced sampling design.

Recommendations

Given the decline in snapper and blue cod numbers within CROP, 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. Inclusion of the two areas along the Leigh coastline not surveyed in 2008 (Areas 1 and 11) should be incorporated in future surveys.

The advent of reef fish monitoring at TMP is a beneficial directive as it provides: 1) necessary baseline information should the TMP become a MPA managed by DoC; 2) important information on marine reserve functioning in the outer Hauraki Gulf, and, 3) due to the consistent methodology, can be compared easily to CROP and other locations that are routinely monitored. Surveys of TMP should be done concurrently with the CROP surveys.

Given the change in fish populations in the recent survey, the effect of reserve protection at CROP and TMP on benthic communities should be assessed. Future fish surveys should be combined with a benthic monitoring program to assess changes in the invertebrate and algae communities.

Continued and frequent monitoring of CROP and TMP is important as MPAs are not static environments and given the potential expansion of coastal development in the Rodney district, the marine environment is likely to be subject to substantial pressures in the future.

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