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**Cape Rodney to Okakari Point Marine Reserve and  
Tawharanui Marine Park Reef Fish Monitoring: UVC Survey  
Autumn 2011**

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Autumn 2011**

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**Tim Haggitt**

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## Executive Summary

- A survey of reef fish diversity and abundance using the Underwater Visual Census technique (UVC) was undertaken within the Cape Rodney to Okakari Point marine reserve (CROP) and Tawharanui Marine Park (TMP) including adjacent unprotected (fished) control areas in autumn 2011. The CROP survey was a continuation of formal monitoring that has taken place since 2000, whereas the survey of TMP was the second survey of this type since its establishment in 2008.
- CROP reef fish assemblage composition in 2011 continued to be distinct from that found in adjacent fished areas, which may, in part, be related to habitat differences between the areas surveyed. Areas surveyed within CROP were almost exclusively dominated by the kelp *Ecklonia radiata* or mixed algal habitat, with non-reserve areas characterised by urchin barrens habitat (the predominant habitat type), mixed algae, and *Ecklonia*.
- Consistent with previous surveys, canonical analysis indicated that butterflyfish, snapper, john dory, banded wrasse, parore, blue cod, and silver drummer were more representative of the reserve sample population with all species attaining higher abundances within CROP in 2011 relative to outside. Higher abundances are likely to reflect higher macroalgal biomass, the absence of fishing, or the combined synergy of both factors. Species diversity was also higher within CROP.
- Reef fish assemblage composition within TMP was not distinct from non-reserve areas, nor did habitats differ between reserve and non-reserve areas to the extent observed within and outside CROP. Despite this, snapper, butterflyfish, red moki, and parore all occurred in higher abundances within TMP in 2011. Species diversity was also higher within TMP compared to outside.
- Snapper abundance in 2011 was higher within CROP at  $4.8 \pm 0.4$  per  $125 \text{ m}^2$  compared to  $1.4 \pm 0.4$  per  $125 \text{ m}^2$  outside. These levels mark an increase in abundance relative to previous surveys, where a linear decline in abundance was evident for both reserve and non-reserve sample populations between 2002 and 2008.
- Snapper abundance was also higher within TMP in 2011 at  $1.78 \pm 0.4 \text{ SE}$  per  $125 \text{ m}^2$  compared to  $0.42 \pm 0.2 \text{ SE}$  per  $125 \text{ m}^2$  outside.
- Comparisons made with 2008 snapper levels indicated much higher abundances of 0+ age (approximately 50 mm Fork Length) and 1+ age (100-150 mm Fork Length) individuals within and outside CROP and TMP in 2011, indicative of strong recruitment over the last few years. There was also a higher frequency of legal-sized snapper recorded within CROP in 2011 compared to outside and compared to 2008 levels.

- Blue cod abundance was also higher within CROP relative to unprotected areas, but levels have remained at  $< 0.2$  per  $125 \text{ m}^2$  since 2008 following a peak in abundance in 2003 ( $\sim 0.5$  per  $125 \text{ m}^2$ ). Periods of low blue cod abundance are thought to be related to elevated sea surface temperatures, which is supported by baited underwater video (BUV) survey data.
- For the present programme, the UVC technique remains useful for making broad comparisons among areas and detecting patterns in fish assemblages through space and time. Undertaking a BUV survey within and outside CROP and TMP in the near future would be recommended to support current UVC results.
- In a recent review the CROP and TMP reef fish survey was considered internationally to be an exceptional example of MPA monitoring due to its design and duration (Addison 2011).
- Presently, marine reserve monitoring in New Zealand is not based on a tangible set of conservation objectives, i.e., national guidelines, standards, and hypotheses that governs MPA research and monitoring. Development of standards would add strength to the current monitoring programme and others of similar nature by providing clear measurable conservation outcomes.
- 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 north-Auckland district, the marine environment is likely to be subject to substantial pressures in the future. Moreover, given that sea surface temperatures are predicted to rise further and that temperature variability can strongly influence reef fish assemblage composition and variability, it will be important to continue monitoring at the current level.

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## 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 (Magnusson 1990; Russ *et al.* 2005; Babcock *et al.* 2010).

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.* 2003a; 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.* 2003a; Shears *et al.* 2006) and through increased biodiversity (Taylor *et al.* 2006). However, recovery processes in marine reserves and associated trophic interactions are complex (e.g., Shears and Babcock 2002, 2003) and may vary considerably among locations (Babcock *et al.* 2010). 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). For example, variation in sea surface temperature, operating on the scale of decades, may strongly influence reef fish assemblages and the geographic range of individual taxa (Masuda 2008).

The Department of Conservation (DOC) has the responsibility for conserving New Zealand's natural and historic heritage for all to enjoy for the present and in the future. 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) at Leigh, 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) 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.* 2003a) and again in 2007 (unpublished data). Tawharanui Marine Park (also a no-take reserve) was established in 1981 and is situated ~ 8 km south of CROP.

The monitoring of marine reserves has three related, but distinctive conservation 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 (e.g., abrupt decline in water quality associated with development versus rising sea surface temperature through time). 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 2011 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.

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.

0+ age-class of recruit snapper (see Francis 1994)

1+ age-class of juvenile snapper 1 year old and greater (see Francis 1994).



## **2.0 Methods**

### **2.1 Survey design**

The 2011 census of the Cape Rodney to Okakari Point Marine Reserve and Tawharanui Marine Park was carried out between April 1 and May 16, 2011. A total of 16 areas incorporating both marine protected areas and adjacent non-protected areas at Leigh Tawharanui and Kawau Island were surveyed (Fig. 2.1). For CROP reserve and adjacent coastline these were Areas 3, 4, 5, 6, 7 & 8 (within the reserve) and Areas 2, 9, 10 & 12 (outside of the reserve). For TMP and adjacent coastline these were Areas 15, 16, 17 (within the reserve) and Areas 18, 19, 20 (outside of the reserve).

The survey design and methods for underwater visual census (UVC) were identical to those used in past surveys (e.g., Taylor *et al.* 2005, Haggitt and Mead 2008). 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 UVC.

### **2.2 Underwater visual census**

Within each site (two per area), two divers surveyed fishes within a total of ten 25 m × 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 on descent. The tape was then 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 fish assemblage data that can be compared to other regions. Occasionally, blue cod and spotties would follow divers between transects, and care was taken not to include those individuals in subsequent replicate transects. Depth at the start and end of each transect and the percent cover occurrence of broad habitat types according to Shears *et al.* (2005) were recorded at the transect origin and at 5m intervals thereafter.

### **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 24 fish species recorded and included in analyses of CROP and TMP and associated control areas. For CROP, observations from Areas 2 and 8 were dropped from the formal analysis, as a balanced dataset was required for the majority of statistical analyses. There was a total of 16 multivariate observations, consisting of 2 stations within each of 8 areas, with 4 areas located inside the reserve (areas 4-7) and 4 areas located outside the reserve (areas 2, 9, 10 and 12). 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).

The null hypothesis for multivariate analysis was that reef fish assemblage composition and variability in 2011 between 1) CROP reserve and equivalent non-reserve areas and 2) TMP and equivalent non-reserve areas were not statistically different from another.

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 the factor “Status” (reserve versus non-reserve) treated as a fixed factor and the factor “Area” treated as a random factor, nested within “Status”. P-values were obtained using appropriate permutation tests (4999 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).

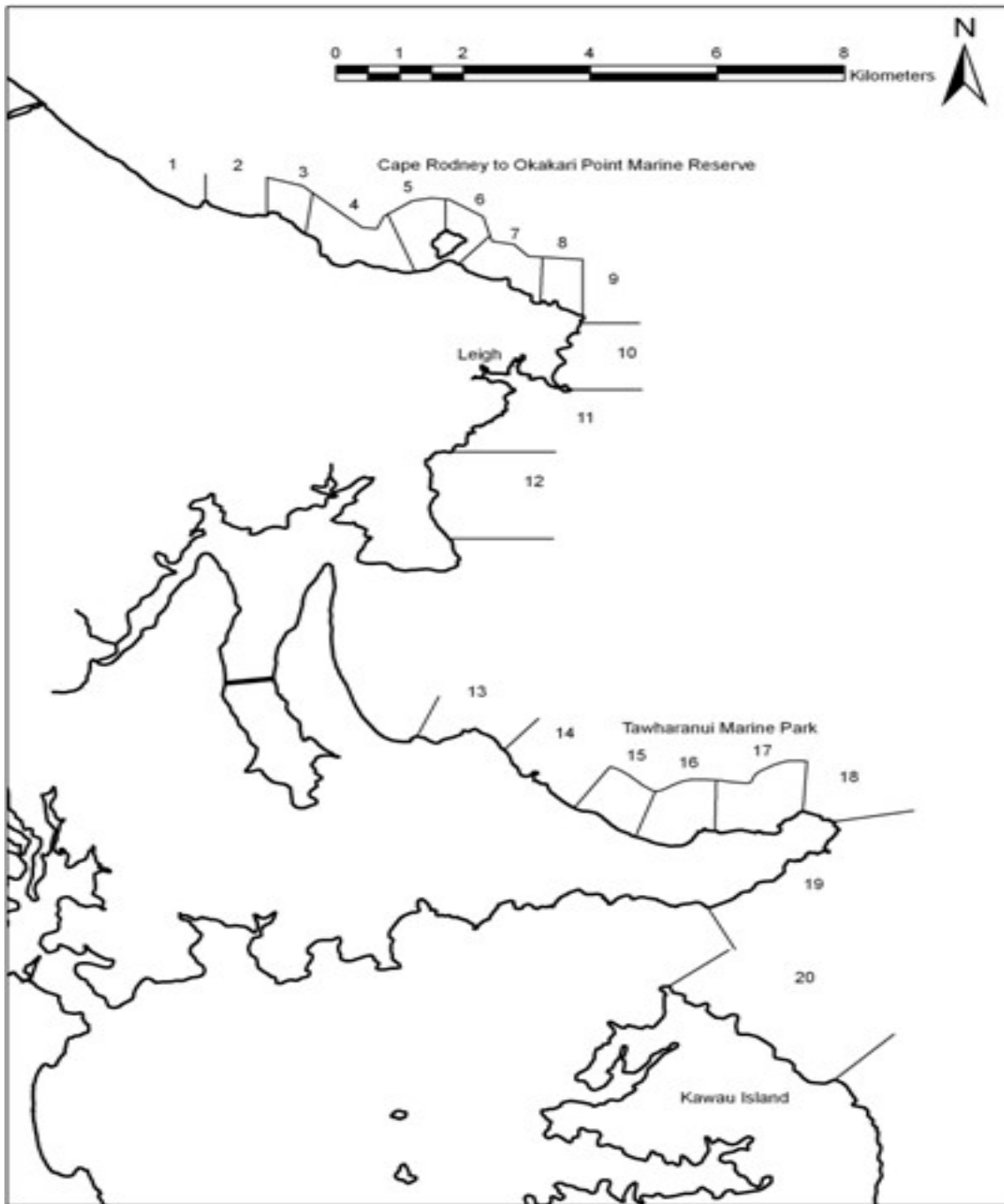
The effect of marine reserve status on reef fish assemblages were also examined using canonical analysis of principal coordinates (CAP, Anderson and Willis 2003, Anderson and Robinson 2003). CAP is a constrained ordination procedure, which finds an axis through the multivariate cloud that is best at discriminating group differences in multivariate space, if differences do exist. CAP is effectively a PCO followed by a traditional canonical discriminant analysis done 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 between reserve and non-reserve areas recorded using UVC were analysed with one-way ANOVA, with “Status” (reserve versus non-reserve) treated as a fixed factor. 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.

To examine contrasts between the means of individual species inside versus those outside each marine reserve, two-sample Wilcoxon rank sum test (equivalent to the Mann-Whitney U test, with continuity correction (Sokal and Rohlf 1981) was used.

The null hypotheses for univariate analysis was that species diversity and individual species abundances in 2011 between CROP reserve and equivalent non-reserve areas and TMP reserve and equivalent non-reserve areas were not statistically different from another.

To help visualise habitat differences among sites based on percent cover, data were square-root transformed and analysed with multidimensional scaling (MDS), based on a Euclidean distance matrix using Primer-E (Clark and Warwick 2001).



**Figure 2.1.** Map of survey areas across the Leigh and Tawharanui regions surveyed in 2011. CROP reserve areas surveyed were 3 to 8, with non-reserve areas 2, 9, 10, and 12. TMP areas surveyed were areas 15 to 17, 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 2011

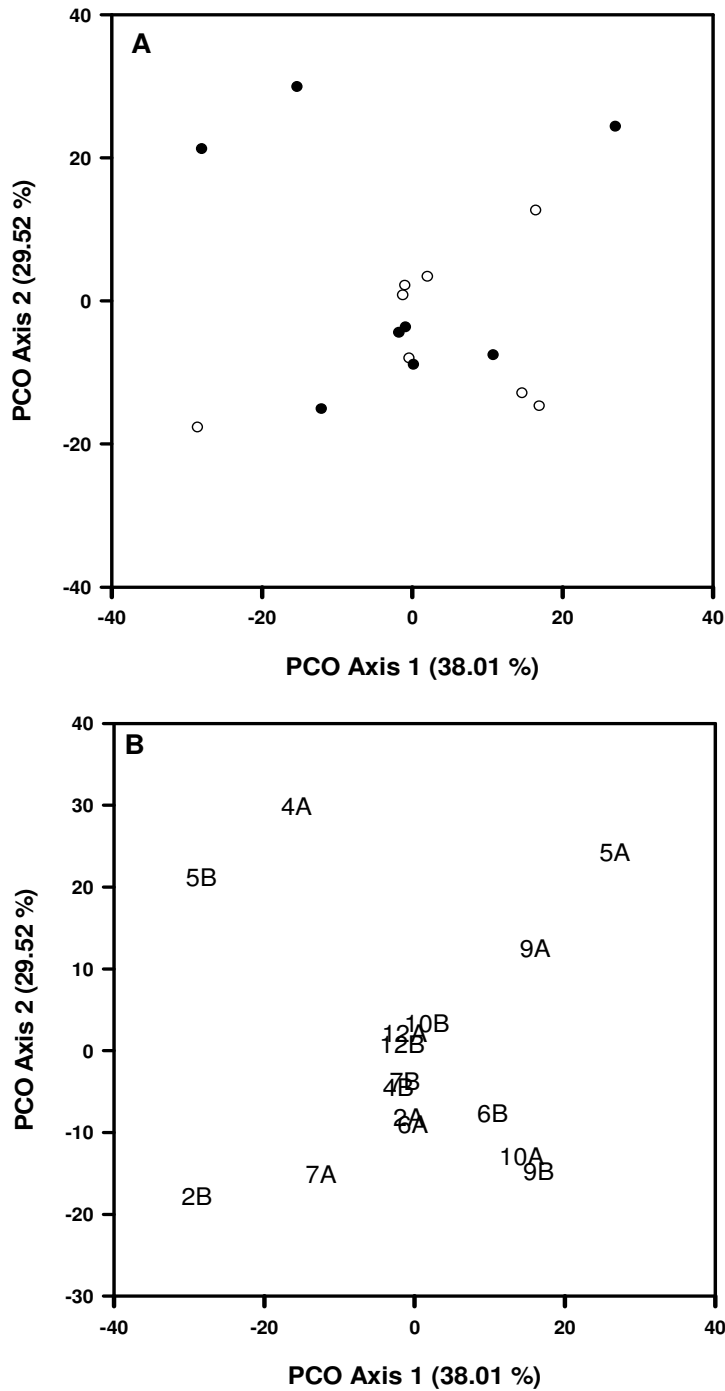
In 2011, the reef fish assemblage composition was statistically different<sup>1</sup> (PERMANOVA – Anderson 2005) between CROP reserve and the non-reserve coastline sampled (Status,  $F = 2.57$ ,  $P < 0.05$ ), although assemblage composition among areas within the reserve and non-reserve (Area (Status))  $F = 1.46$ ,  $P = 0.074$ ) was not statistically significant. There was no statistically significant difference in the multivariate dispersion of fish assemblages between the reserve and non-reserve areas (PERMDISP Anderson 2004). This is supported by the PCO plot (metric MDS) which shows a wide spread of reserve and non-reserve sampling sites (Fig. 3.1) across the ordination.

Canonical analysis (CAP) indicated differences between assemblages inside versus outside of the reserve with distinct separation (no overlap) between the two groups (Fig. 3.2). The squared canonical correlation was 0.77 and the discriminant function correctly allocated 7 out of 8 non-reserve stations and 7 out of 8 reserve stations with a total allocation success of 87.5 %. The permutation test indicated that the level of discrimination was statistically significant ( $\delta^2 = 0.73$ ,  $P = 0.001$ ). A range of species had higher frequencies and were accordingly more-representative of fish assemblages within the reserve including butterflyfish, snapper, john dory, banded wrasse, blue cod and silver drummer (Table 3.1). Assemblages outside of the reserve had higher frequencies of sweep, spotty, and goatfish (Table 3.1).

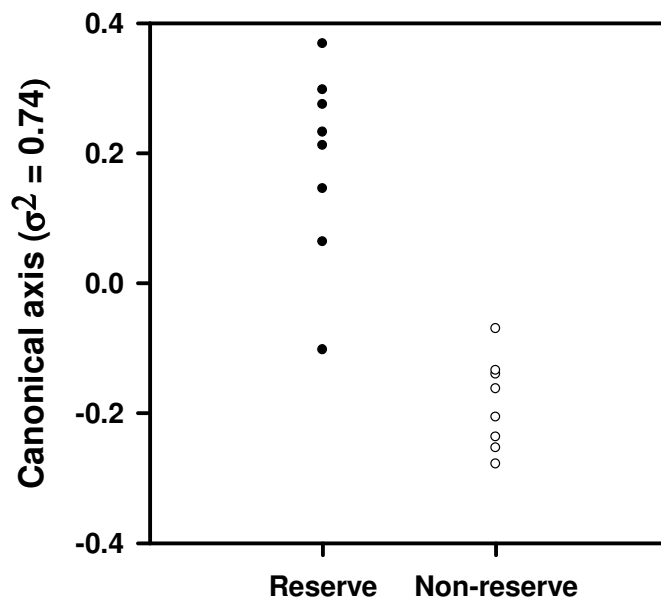
**Table 3.1.** Individual species having correlations of  $|r| > 0.20$  with the canonical axis separating reserve from non-reserve sites and occurring in at least 10% of the sites.

Positive correlation (reserve)		<i>r</i>
Butterfish	<i>Odax pullus</i>	0.601
John dory	<i>Zeus faber</i>	0.523
Snapper	<i>Pagrus auratus</i>	0.565
Silver drummer	<i>Odax pullus</i>	0.378
Banded wrasse	<i>Notolabrus fucicola</i>	0.284
Red moki	<i>Cheilodactylus spectabilis</i>	0.367
Blue cod	<i>Parapercis colias</i>	0.334
Negative correlation (non-reserve)		
Sweep	<i>Scorpius lineolatus</i>	-0.540
Spotty	<i>Notolabrus celidotus</i>	-0.449
Goatfish	<i>Upeneichthys lineatus</i>	-0.406

<sup>1</sup> To create a balanced dataset statistical analysis was undertaken using reserve sites 4-7 and non-reserve sites 2,9,10 and 12, for PCO, PERMANOVA, PERMDISP and CAP analysis,.



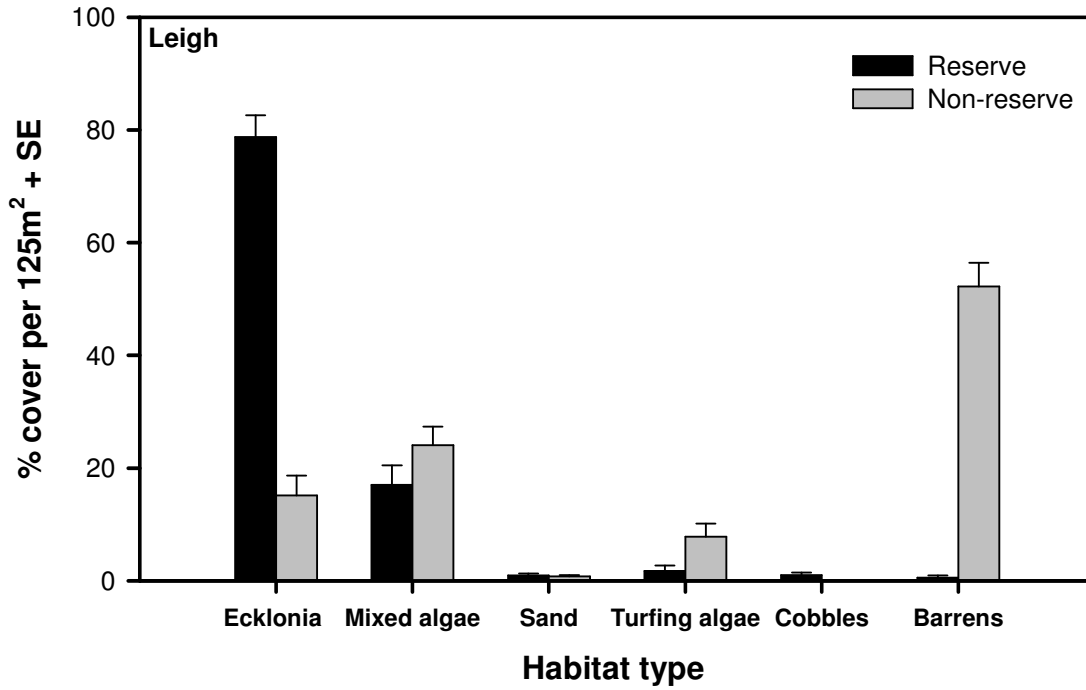
**Figure. 3.1.** Ordination plot of the first two PCO axes (explaining 82.7 % of the original variability) based on Bray-Curtis dissimilarities of  $\ln(y+1)$  transformed species abundance data (24 species). The plots depict assemblages at different stations based on status for (A) CROP reserve (dark symbols) versus non-reserve (open symbols) and (B) based on area, with reserve areas 4 through 7 (with 2 stations A,B per area) and non-reserve areas 3, 8,9,10 and 12 (2 stations per area A, B).



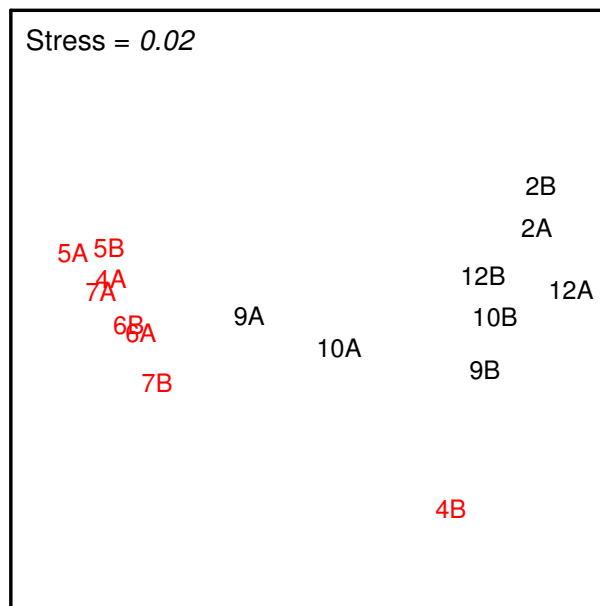
**Figure 3.2.** Plot of the canonical axis from a CAP constrained ordination to discriminate fish assemblages from CROP reserve versus non-reserve stations. The discriminant analysis was done on the first  $m = 8$  PCO axes (which explained 99.99% of the original variability) from Bray-Curtis dissimilarities of  $\ln(y+1)$  transformed species abundances (24 species).

On average there were fewer species observed in areas outside the reserve (mean =  $10.8 \pm 2.5$  SE), compared to areas inside the reserve (mean =  $12.3 \pm 1.1$  SE) but this difference was not statistically significant ( $F_{1,8} = 1.06$ ,  $P = 0.306$ ).

As suggested for previous surveys (Taylor *et al.* 2005, Haggitt and Mead 2008), differences in reef fish assemblage composition between CROP reserve and non-reserve areas may be related to the prevalence of different habitat types across sample transects, with greater habitat variability occurring outside. Based on percent habitat occurrence, the kelp *Ecklonia radiata* (hereafter *Ecklonia*) was by far the most dominant habitat type across reserve areas surveyed, followed by mixed algae (Fig. 3.3). Together *Ecklonia* and mixed algae accounted for over 90 % of the habitat types recorded. Urchin barrens, turfing algae, cobbles and sand were all < 2 %) (Fig. 3.3). Conversely, urchin barrens was the dominant habitat type (> 50 %) in non-reserve areas followed by mixed algae, *Ecklonia*, turfing algae, and sand (Fig. 3.3). Habitat types presented as an MDS ordination (Fig. 3.4) illustrates a clear division between reserve and non-reserve sites, with reserve sites predominantly located to the left of the ordination and non-reserve predominantly to the right.



**Figure 3.3.** Main habitat types based on percent cover (pooled across areas) for CROP reserve and non-reserve locations surveyed in 2011.



**Figure 3.4.** MDS plot based on 6 habitat classes, for CROP reserve sites (red) and non-reserve sites (black) surveyed in 2011.

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

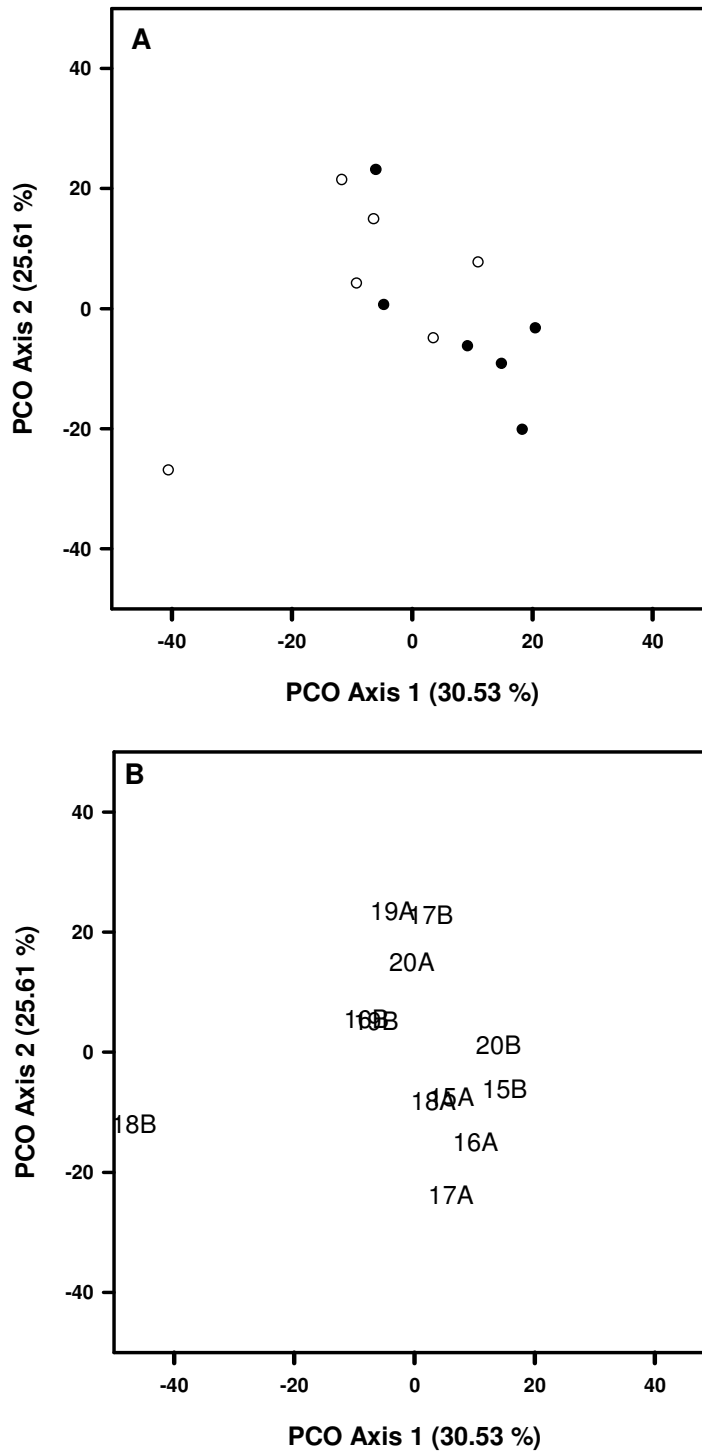
Reef fish assemblage composition between TMP reserve and non-reserve areas in 2011 was not statistically significant (PERMANOVA) (Status  $F = 1.54$ ,  $P = 0.129$ ), nor were the individual areas within TMP and the non-reserve coastline (Area (Status),  $F = 1.11$ ,  $P = 0.34$ ). This is reflected in the PCO ordination with no clear grouping or separation between reserve and non-reserve sites surveyed (Fig. 3.5). Further, multivariate dispersion analysis (PERMDIS – Anderson 2004) indicated no statistically significant difference in assemblage variability ( $F = 0.11$ ,  $P = 0.76$ ) between reserve and non-reserve areas surveyed.

Additional canonical analysis, used to examine differences between fish assemblages inside the reserve versus outside was not compelling. The squared canonical correlation was 0.27 and there was clear overlap between the two groups in the discriminant model (Fig. 3.6). The permutation test found the level of discrimination was not statistically significant (Fig. 3.6, canonical correlation,  $\delta^2 = 0.234$ ,  $P = 0.533$ ).

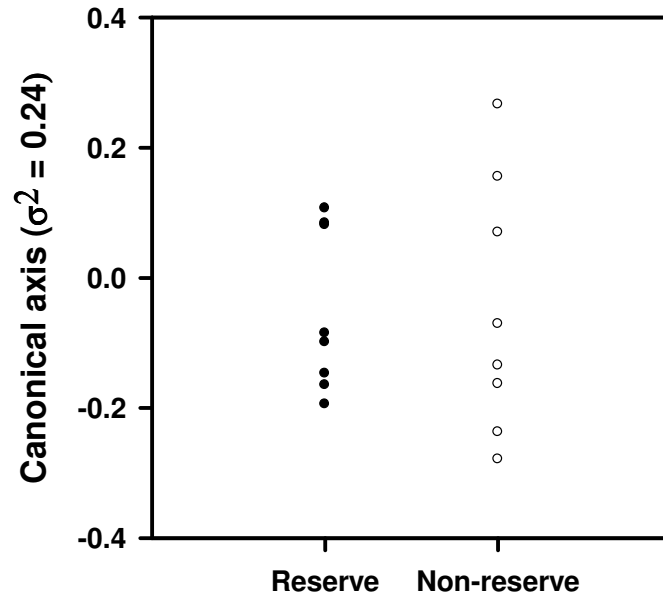
Species diversity was higher inside the reserve (mean =  $11.8 \pm 0.8$  SE), relative to areas outside (mean =  $8.83 \pm 1.1$  SE), and this difference was statistically significant ( $F_{1,10} = 5.23$ ,  $P = 0.0452$ ).

Habitats with the highest mean percent cover within TMP were mixed algal habitat *Ecklonia* forest and urchin barrens, with sand and cobbles comprising < 4 % (Fig. 3.7). A similar pattern was apparent for non-reserve areas with mixed algae and *Ecklonia* the two most dominant habitat types (> 30 %), followed by urchin barrens habitat (~ 25 %), with remaining habitats < 4 %. The percent cover of barrens within TMP was also much higher than that enumerated within CROP (Fig. 3.7). The MDS ordination highlights further the general accordance of habitats between reserve and non-reserve sites with no obvious separation between groups (Fig. 3.8).

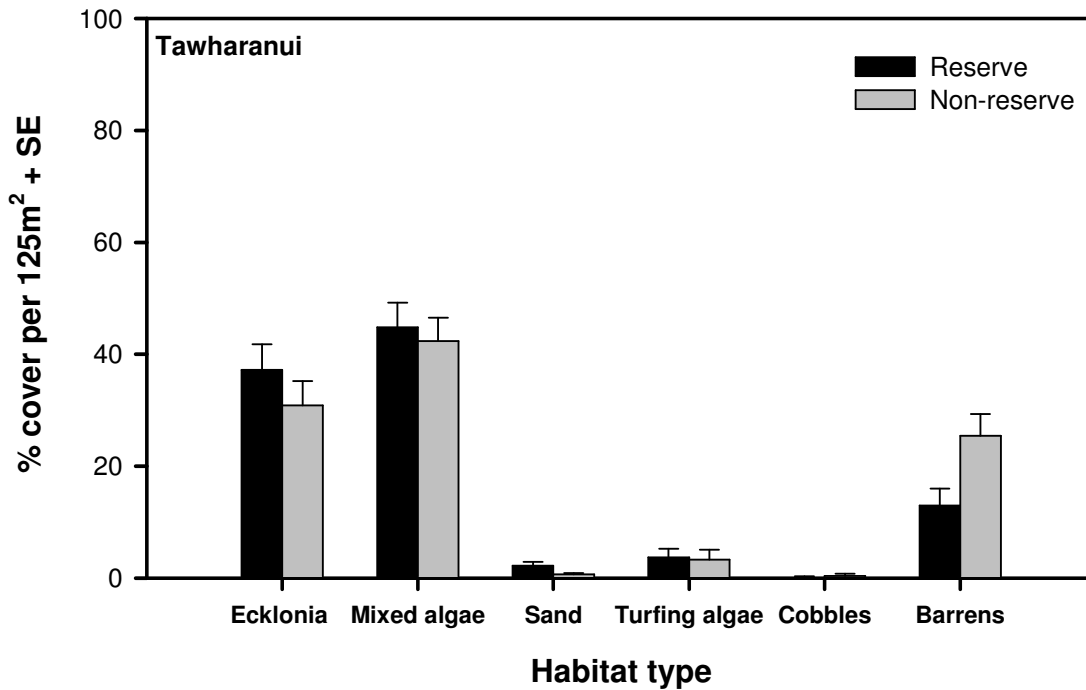




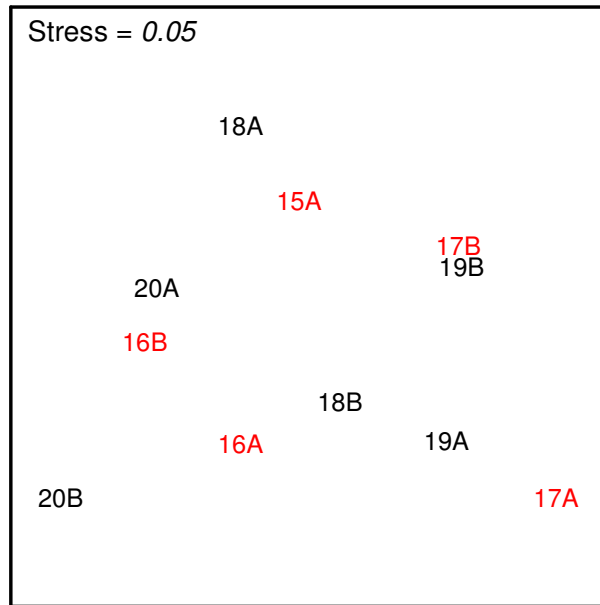
**Figure 3.5.** Ordination plot of the first two PCO axes (explaining 56.14% of the original variability) based on Bray-Curtis dissimilarities of  $\ln(y+1)$  transformed species abundance data (24 species). The plots depict assemblages at different sites with labels based on status for (A) TMP (dark symbols) versus non-reserve (open symbols) status and (B) based on area for TMP reserve areas 15 through 17 (with 2 stations A,B per area) and non-reserve areas 18 through 20.



**Figure 3.6.** 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 (24 species).



**Figure 3.7.** Main habitat types based on percent cover (pooled across areas) for TMP reserve and non-reserve locations surveyed in 2011.



**Figure 3.8.** MDS plot based on 6 habitat classes, for TMP sites (red) and non-reserve sites (black) surveyed in 2011.

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

A consistent trend for UVC surveys undertaken for CROP reserve and the corresponding non-reserve coastline has been high variability through time for a range of taxa, particularly pelagic species (e.g., blue maomao, sweep, kahawai etc). While this was true for the 2011 survey, several trends were also apparent. Abundance levels for a range of common fish species are presented in Figs 3.9; 3.12-3.16.

In 2011, snapper (*Pagrus auratus*) abundance within the reserve was the highest recorded since the inception of the monitoring programme at  $4.8 \pm 0.4$  SE per  $125 \text{ m}^2$  and statistically different ( $Z = 6.59$ ,  $P < 0.0001$ ) from non-reserve levels at  $1.4 \pm 0.4$  SE per  $125 \text{ m}^2$ . Interestingly, non-reserve snapper abundance was the highest recorded since 2002. The 2011 levels contrasts with the linear decline in abundance evident within reserve and non-reserve sample populations between 2002 and 2008 (Fig. 3.9)<sup>2</sup>. Within the reserve, highest snapper abundance occurred in the two central areas (4 and 5) with a progressive decline in abundance with increasing distance to the north-eastern (Area 3) and south-western (Area 8) reserve boundaries (Fig 3.10), a trend that matches past BUV surveys at CROP and Hahei (Taylor *et al.* 2005, 2006)<sup>3</sup>.

<sup>2</sup> Note: Both 2008 and 2011 levels are based on 4 areas being surveyed within and outside the reserve whereas pre-2008 surveys are based on 6 areas.

<sup>3</sup> Data from Areas 2 and 8 within CROP are presented for Fig. 3.10; however these sites are not included in formal statistical analyses.

Snapper were ~ 2-fold larger within the reserve ( $270.4 \text{ mm} \pm 16.5$  (95 % CI)), compared to the non-reserve sample population ( $129.6 \text{ mm} \pm 13.7$  (95 % CI) non-reserve) and this difference was statistically significant ( $Z = 8.09$ ,  $P < 0.0001$ ). Due to higher numbers of snapper being counted and sized within CROP and across the non-reserve coastline in 2011, size frequency distributions differ considerably to those of 2008 (Fig. 3.11). For CROP this is visible in an increase of small sub-legal individuals particularly those  $\leq 100$  mm FL (corresponding to 0+ and 1+ age classes), coupled with a higher frequency of individuals between 300-400 mm FL and larger snapper  $> 500$  mm FL (Fig. 3.11). Similarly, for the non-reserve sample population there was a greater frequency of individuals of FL  $\leq 100$  mm relative to other size groupings and in relation to 2008 levels. Legal-sized snapper were also more abundant relative to 2008 levels; although UVC may underestimate non-reserve legal-sized *P. auratus* abundance, due to their tendency to display “diver negative” behaviour.

Consistent with previous surveys, blue cod abundance in 2011 was higher inside CROP compared to outside, with 2011 levels similar to the 2002 autumn survey, i.e.,  $< 0.15$  individuals per  $125 \text{ m}^2$  (Fig. 3. 9). The current abundance marks the lowest recorded since a 2003 peak in abundance of around 0.4 individuals per  $125 \text{ m}^2$ , which was followed by a steady decline. In past BUUV and UVC surveys increased sea surface temperature has been touted as playing a causative role in the low numbers of blue cod numbers across the Leigh area (Taylor *et al.* 2005).

A range of species had higher abundances within CROP relative to the unprotected Leigh coastline in 2011, and, for the most part, exhibited broad commonality with abundance levels estimated in 2008. These included red moki (*Cheilodactylus spectabilis*), banded wrasse (*Notolabrus fucicola*), parore (*Girella tricuspidata*), silver drummer (*Kyphosus sydneyanus*), blue maomao (*Scorpiis violaceus*) and butterflyfish (*Odax pullus*). Of these, red moki abundance has remained reasonably stable between 2005 and 2011 at  $> 1$  individual per  $125 \text{ m}^2$  with non-reserve abundance consistently  $< 1$  individual per  $125 \text{ m}^2$  (Fig. 3.9) with the difference in 2011 statistically significant ( $Z = 3.29$ ,  $P < 0.001$ ). Butterflyfish remain at  $> 0.5$  individuals per  $125 \text{ m}^2$ , a level maintained since 2003, with non-reserve levels remaining at  $< 0.2$  individuals per  $125 \text{ m}^2$  (Fig. 3.16). The difference between reserve and non-reserve levels in 2011 was also statistically significant ( $Z = 3.29$ ,  $P < 0.001$ ).

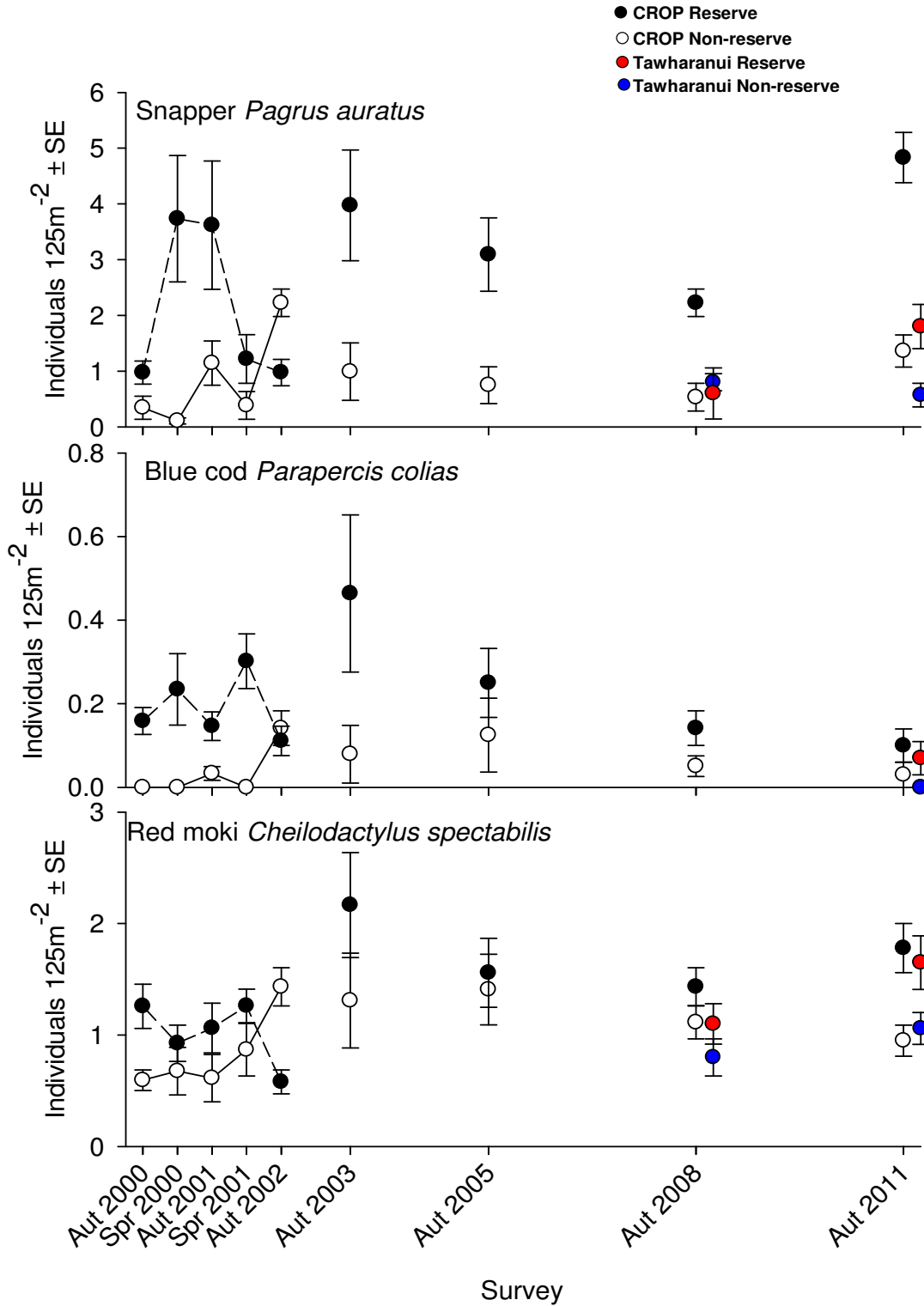
Parore numbers steadily increased within the reserve between 2002 and 2008 from around 0.5 per  $125 \text{ m}^2$  to 1.5 per  $125 \text{ m}^2$ , with levels in 2011 in accordance with 2008 (Fig 3.13). With the exception of 2005, non-reserve levels have consistently remained  $< 0.5$  individuals per  $125 \text{ m}^2$  since the inception of sampling in 2008. Differences between reserve and non-reserve levels in 2011 were statistically significant ( $Z = 2.59$ ,  $P < 0.001$ ).

Silver drummer maintained higher abundances within the reserve relative to outside in 2011 and the difference was statistically significant ( $Z = 2.98$ ,  $P < 0.001$ ). Principally, silver drummer levels have remained  $> 1$  individual per  $125 \text{ m}^2$  since the first survey, with peaks in abundance at around 2 individual per  $125 \text{ m}^2$  in 2000 and 2005 (Fig. 3.14).

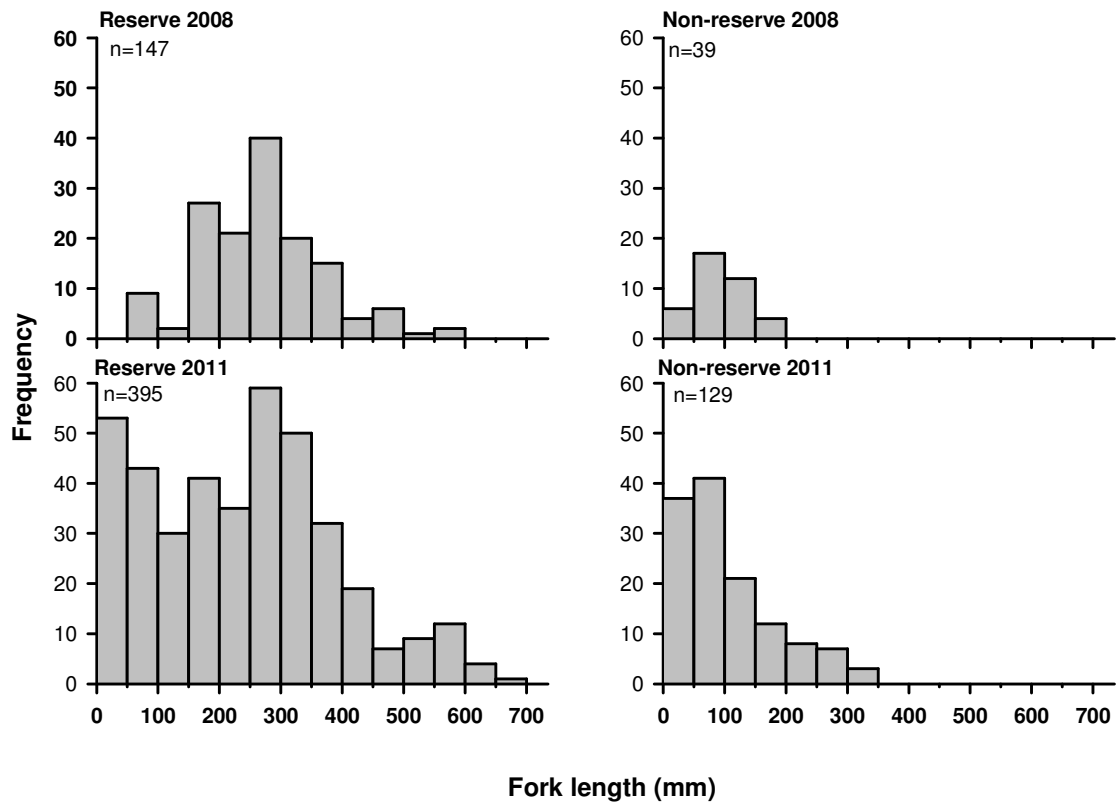
Conversely the non-reserve sample population has remained at  $< 0.5$  individuals per  $125 \text{ m}^2$  across surveys.

The only species that has continually displayed higher abundance in non-reserve areas through time is the goatfish (*Upeneichthys lineatus*) (Fig. 3.14). While the current level of 2.19 individual per  $125 \text{ m}^2$  marks a slight decline from 2008 levels, abundance still remains  $> 2$ -fold that of the reserve and statistically different ( $Z = -2.14, P < 0.001$ ).

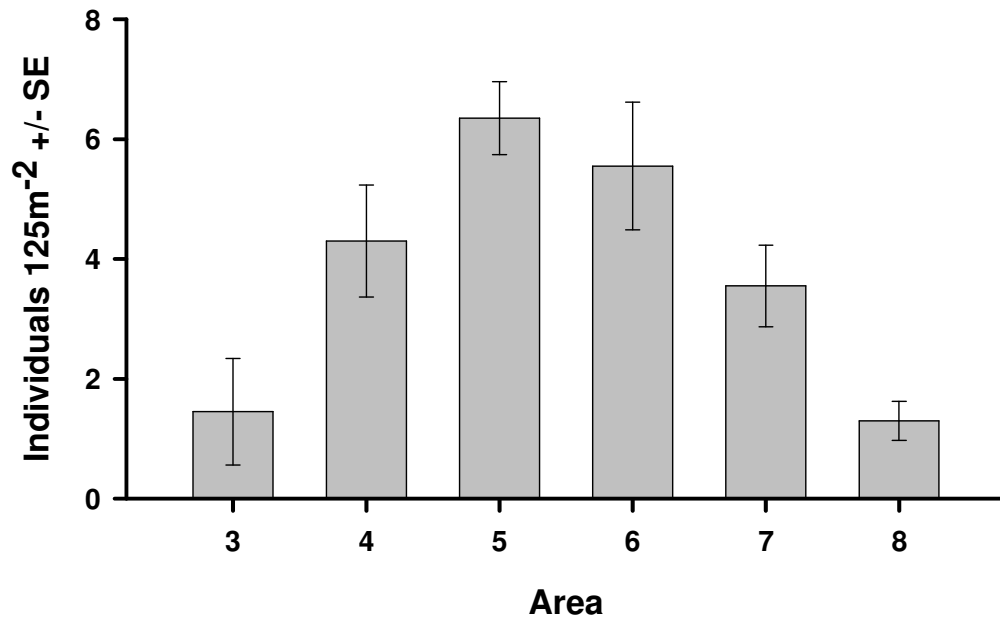
Abundance patterns of schooling taxa such as kahawai (Fig. 3.13), jack mackerel (Fig. 3.13), and sweep (Fig. 3.15) have remained highly variable across reserve and non-reserve areas from the beginning of the monitoring programme and again were highly variable in 2011. Interpretation of trends through time for these species is therefore difficult.



**Figure 3.9.** 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 2011 and inside and outside Tawharanui Marine Park between 2008 and 2011. Note: y axis scale differs among plots.

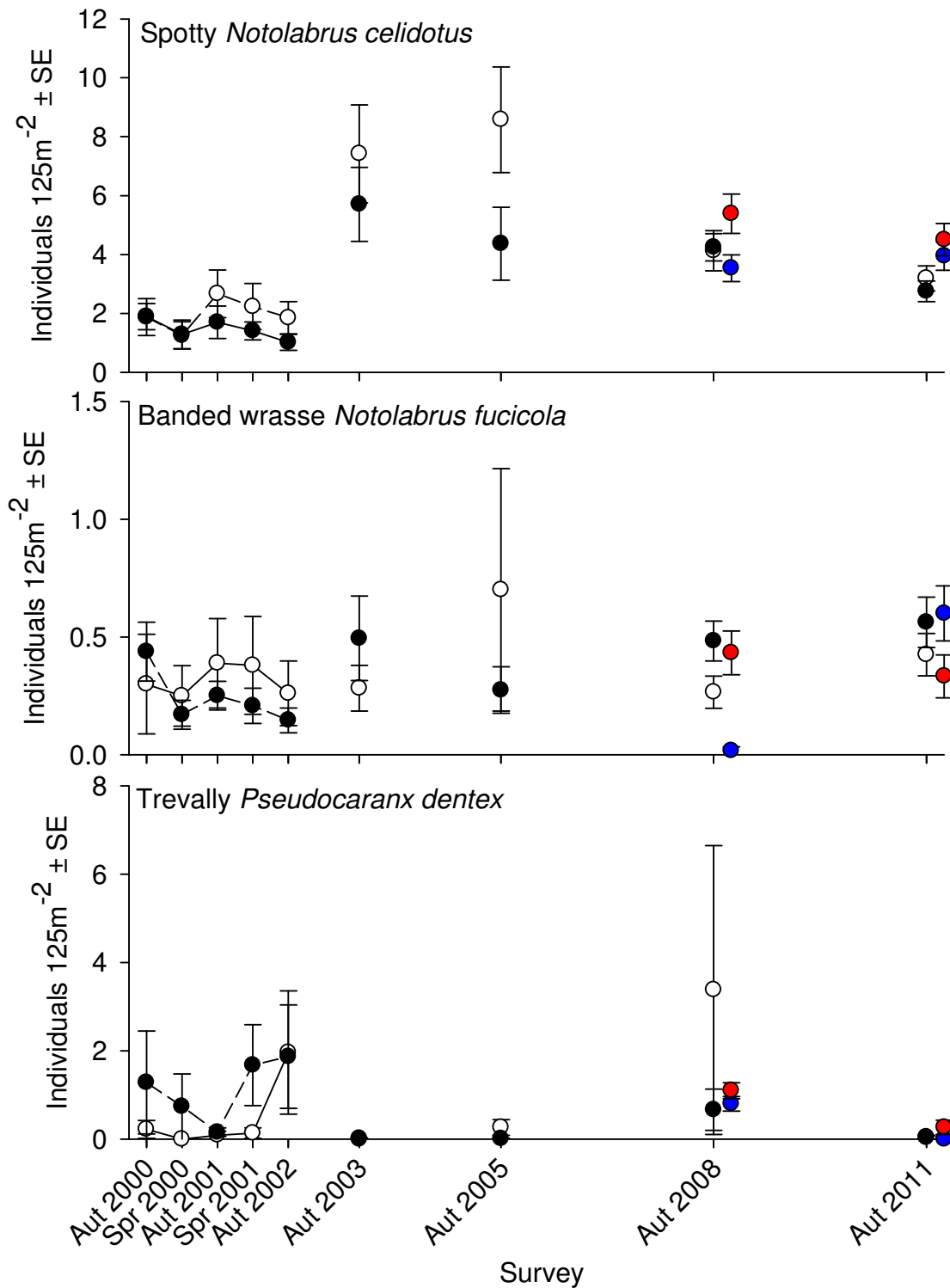


**Figure. 3.10** Size frequency distributions for snapper (*Pagrus auratus*) within CROP reserve and non-reserve areas surveyed for 2008 and 2011. Data are pooled across respective survey areas.

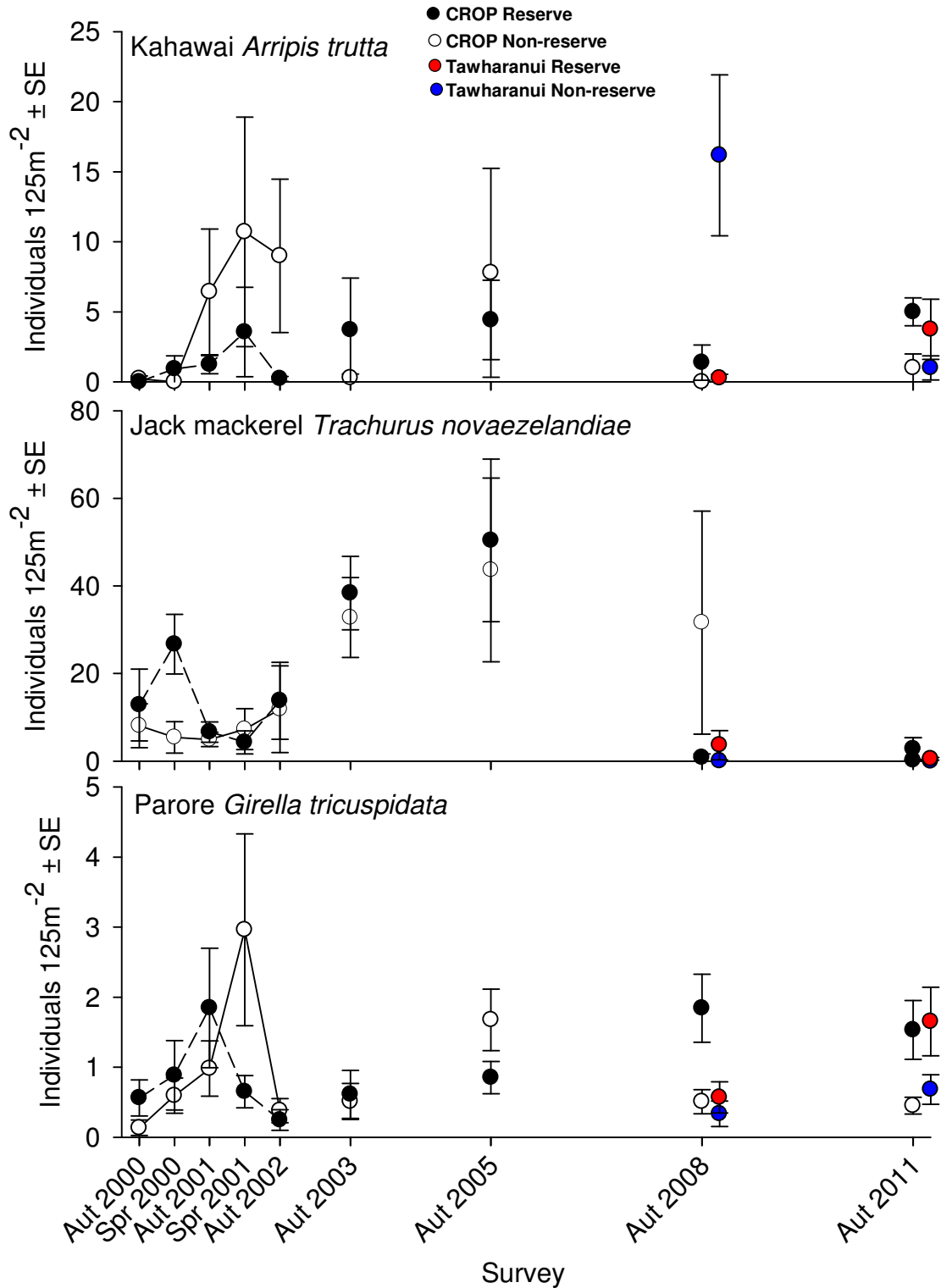


**Figure. 3.11.** Abundance levels of snapper (*Pagrus auratus*) in the six areas (3-8) within CROP reserve (refer to Fig. 2.1) in 2011. Data are pooled across 2 sampling sites within each area.

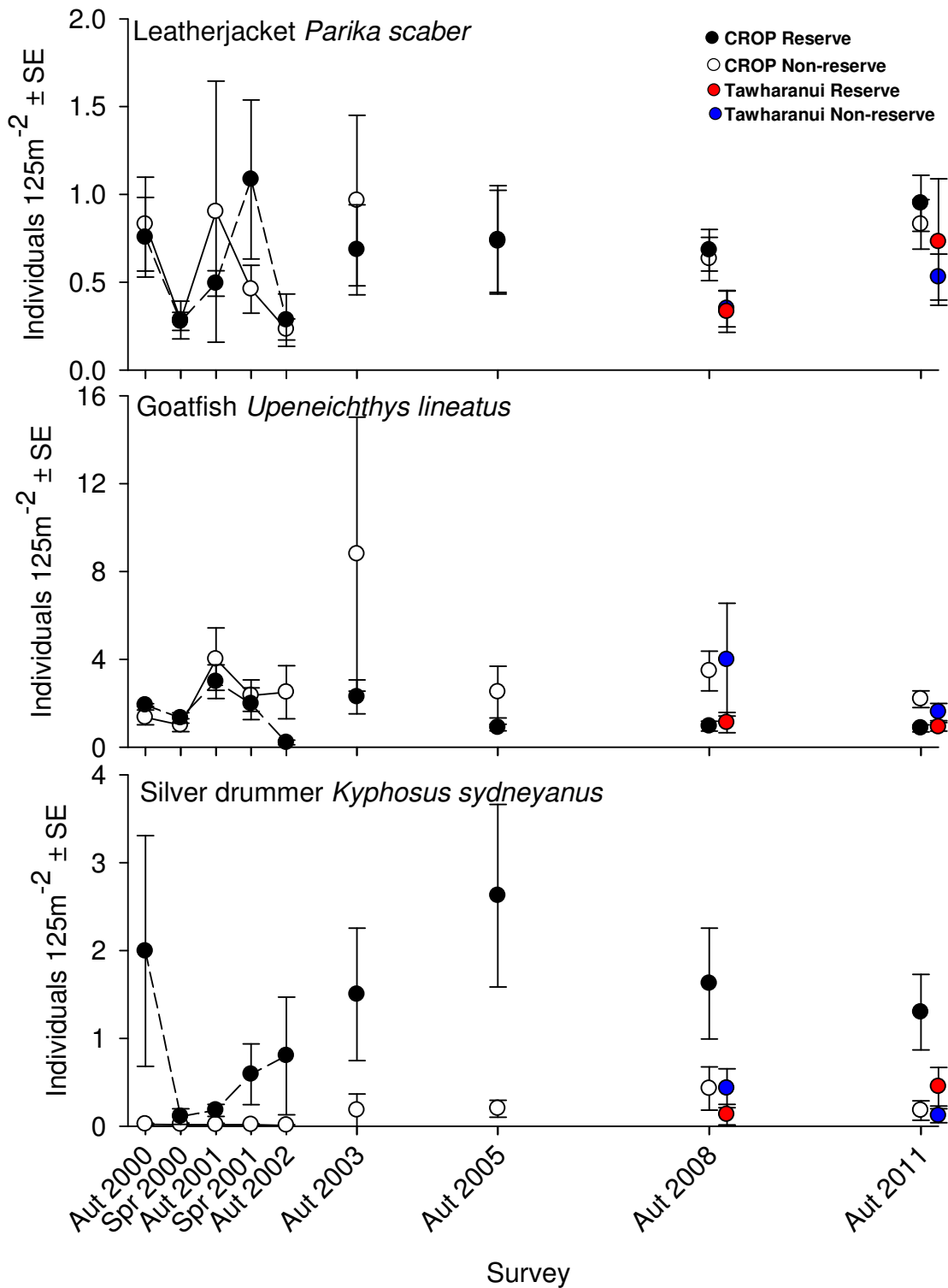




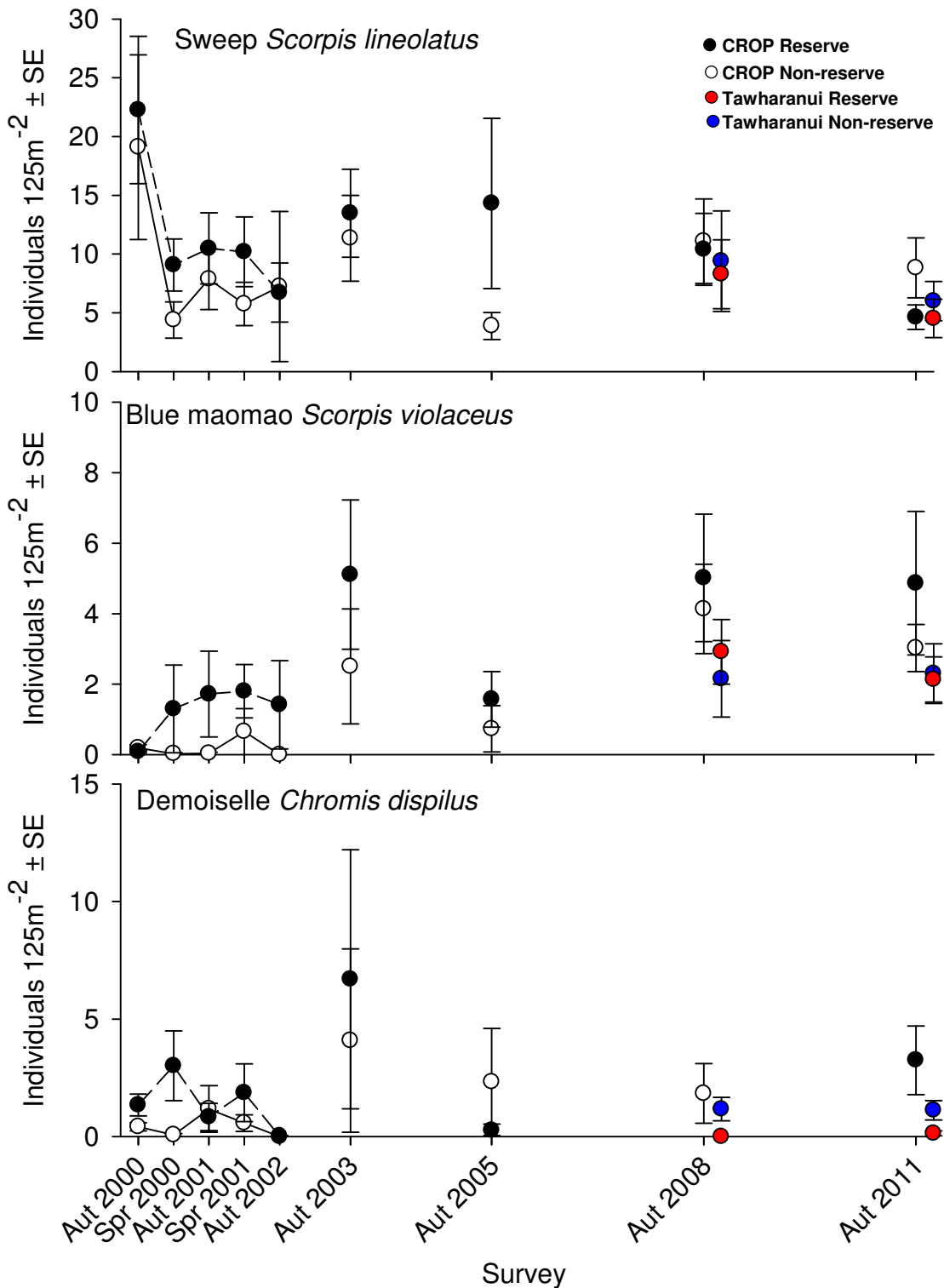
**Figure 3.12.** Long term trends in the densities of spotty, banded wrasse, and trevally inside and outside the Cape Rodney to Okakari Point Marine Reserve, as measured using UVC between 2000 and 2011 and inside and outside Tawharanui Marine Park between 2008 and 2011 Note: y axis scale differs among plots.



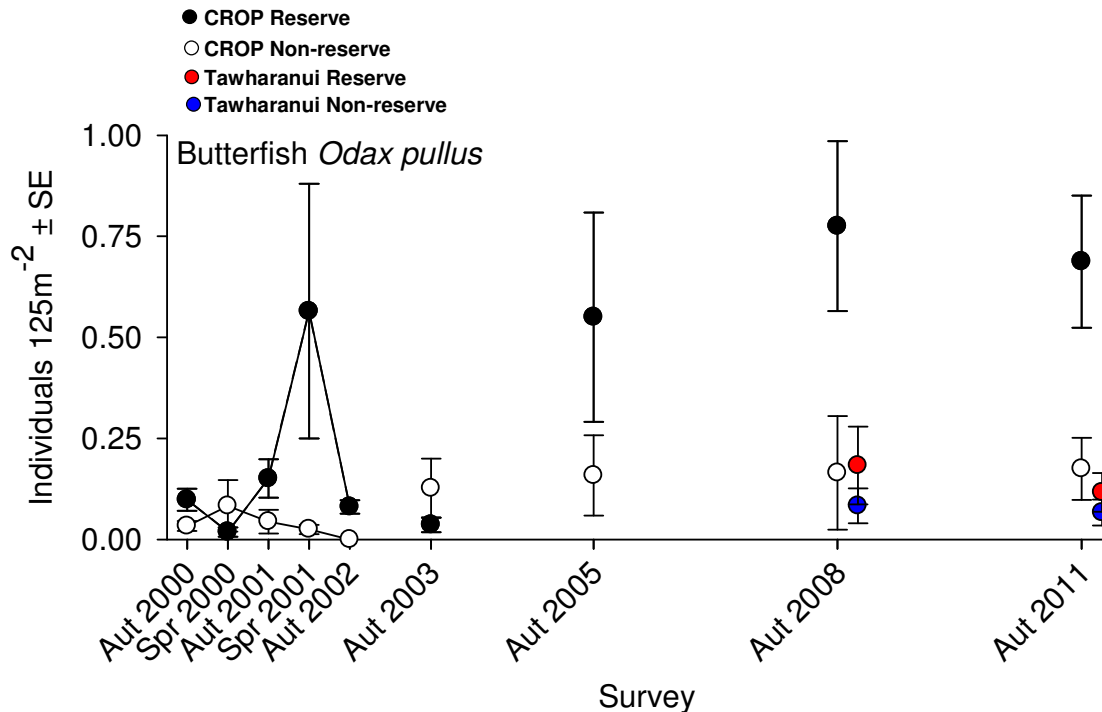
**Figure 3.13.** Long term trends in the densities of kahawai, jack mackerel, and parore inside and outside the Cape Rodney to Okakari Point Marine Reserve, as measured using UVC between 2000 and 2011 and inside and outside Tawharanui Marine Park between 200 and 2011. Note: y axis scale differs among plots.



**Figure 3.14.** 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 2011 and inside and outside Tawharanui Marine Park between 2008 and 2011. Note: y axis scale differs among plots.



**Figure 3.15.** 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 2011 and inside and outside Tawharanui Marine Park between 2008 and 2011. Note: y axis scale differs among plots.



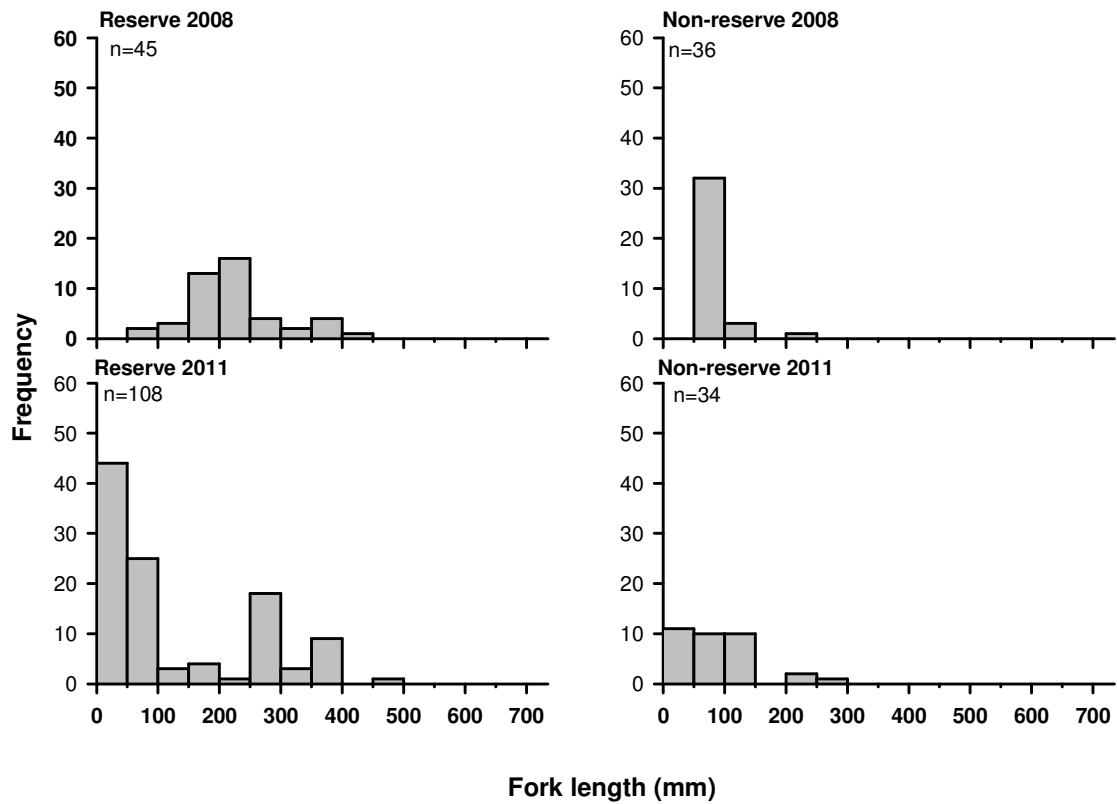
**Figure 3.16.** Long term trends in the density of butterflyfish inside and outside the Cape Rodney to Okakari Point Marine Reserve, as measured using UVC between 2000 and 20011 and inside and outside Tawharanui Marine Park between 2008 and 2011.

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

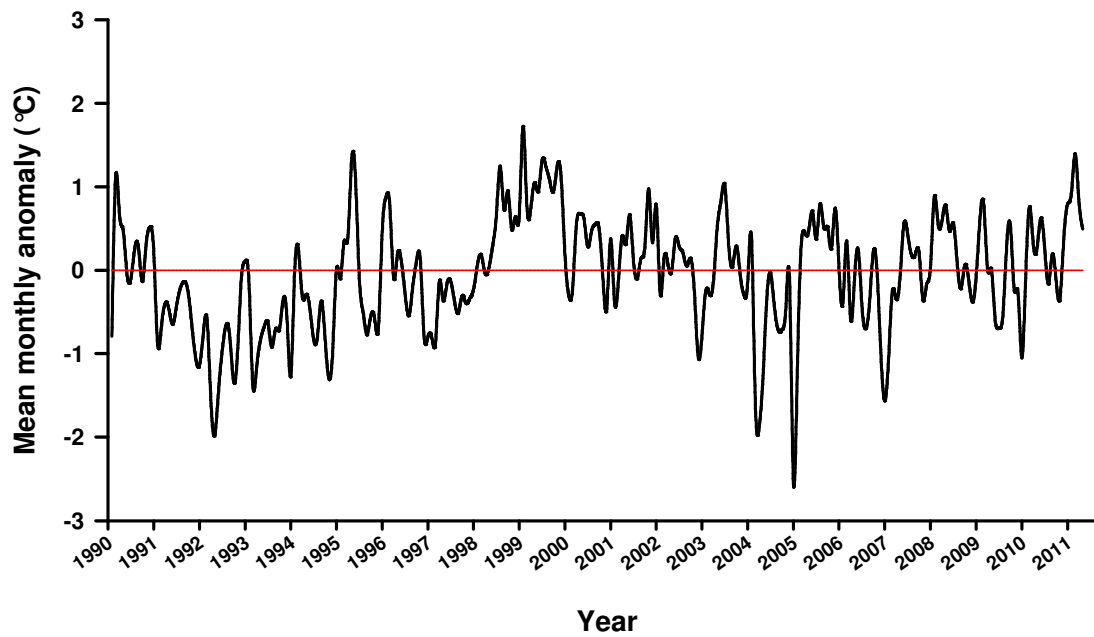
Mean snapper abundance in 2011 was higher within TMP ( $1.78 \pm 0.4$  SE per  $125 \text{ m}^2$ ) compared to outside ( $0.42 \pm 0.2$  SE per  $125 \text{ m}^2$ ) (Fig. 3.9) and this difference was statistically significant ( $Z = 3.83$ ,  $P < 0.0001$ ). Snapper mean size was also larger for the reserve snapper sample population ( $154.35 \text{ mm} \pm 23.7$  (95 % CI) compared to the non-reserve sample population ( $113.23 \text{ mm} \pm 21.2$  (95 % CI), but this difference was not statistically significant ( $Z = 4.15$ ,  $P = 0.67$ ). Juvenile snapper  $< 150 \text{ mm FL}$  (0+ and 1+ age classes), were a feature of both sample populations (Fig. 3.17), although within the reserve a much higher frequency of both sub-legal and legal-sized snapper were apparent. Several large schools of juvenile snapper were however observed outside of sample transects in non-reserve areas at Kawau Island.

As the survey of TMP in 2011 was the second consecutive survey, temporal patterns cannot be evaluated to the same extent as for CROP. Fish species (other than snapper) recorded in higher abundances within the reserve and statistically different ( $P < 0.05$ ) from non-reserve abundances in 2011 included red moki, blue cod, leather jacket, parore butterflyfish and silver drummer. These species all increased in abundance between surveys. Species such as banded wrasse and goatfish demonstrated higher abundances

outside the reserve with banded wrasse in particular increasing dramatically in abundance between surveys (Figs 3.9; 3.12-3.16).



**Fig. 3.17.** Size frequency distributions for snapper (*Pagrus auratus*) within TMP reserve and non-reserve areas for 2008 and 2011. Data are pooled across respective survey areas.



**Figure 3.18.** Sea surface temperature anomalies from Leigh between 1990 and 2011 (based on long term average 1967-2007). La Niña conditions are typically associated with +ve values and El Niño with -ve values.

## 4.0 Discussion

The 2011 reef fish survey of CROP was the ninth survey since 2000 and the second such survey for TMP since its amalgamation into the programme in 2008. Long-term monitoring of this nature is particularly important in assessing the nature and rate of recovery/changes of MPAs in response to protection, i.e., identification of time lags between cause (protection) and subsequent effects (increase in exploited species, habitat change etc) (Magnusson 1990; Babcock *et al.* 2010) against natural variation through time. Further, the consistency of the reef fish survey design employed here allows direct comparison to other MPAs that are routinely surveyed for reef fish abundance in north-eastern New Zealand such as Hahei (Taylor *et al.* 2006) and the Poor Knights Islands (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).

Reef fish assemblage composition and variability within CROP and TMP including non-reserve areas, for the most-part, exhibited commonality with the previous 2008 survey (see Haggitt and Mead 2008). This included higher species diversity within CROP relative to outside and differences between reserve and non-reserve assemblages. Higher diversity within TMP relative to outside was also detected, although reserve and non-reserve reef fish assemblages were similar to each other.

For CROP, the difference between reserve and non-reserve reef-fish assemblages, to a certain extent, is likely due to habitat differences between the two areas (see Shears and Babcock 2003). 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; Haggitt and Mead 2008) and the influence of habitat has been demonstrated to affect reef fish assemblages in purposely designed studies (see Anderson and Millar 2004; Williams *et al.* 2008). CROP reserve areas sampled in 2011 continue to be dominated by macroalgae, particularly *Ecklonia* at depths > 6m and mixed algal habitat < 6m (together accounting for over 95% of the habitat cover), whereas the majority of non-reserve sites sampled were a combination of urchin barrens (the dominant habitat type), mixed macroalgae and *Ecklonia* forest. Within CROP, urchin barrens was either completely absent, or where it was observed, severely protracted relative to non-reserve areas.

Canonical analysis indicated that butterfish, red moki, snapper, john dory, banded wrasse, blue cod and silver drummer were responsible for the separation between reserve CROP reserve and non-reserve areas, with all species having higher abundances within the reserve; a pattern also evident in past surveys (Taylor *et al.* 2005; Haggitt and Mead 2008). Higher butterfish abundance within CROP through time is possibly due to two factors. First the higher biomass of *Ecklonia* and *Carpophyllum* within the - two main food sources of this taxon (Clements and Choat 1993) – is likely to positively influence butterfish abundance and; second, the absence of fishing pressure within the reserve, as butterfish remain a focal target for spearfishers. Spearfishing is an activity that has increased in popularity in recent years around the Leigh coastline with red moki and john



dory also targeted by this activity (personal observation). Higher abundances of parore and silver drummer may also be reflective of greater macroalgal biomass within CROP, as *Ecklonia*, *Carpophyllum* and associated filamentous algae are key components of the diets of these two species (Choat and Clements 1992; Moran and Clements 2002).

The lack of any difference in reef fish composition and variability between TMP and corresponding non-reserve areas is potentially related to less habitat variation between the two areas. For example, urchin barrens habitat, a prevalent habitat type at many non-reserve sites, also occurred at half of TMP sites surveyed. In addition, mixed algal habitat, the dominant habitat within TMP, has also increased outside the reserve in recent years in tandem with a decline in the extent of urchin barrens (N. Shears personal communication in 2011). Despite the lack of assemblage-level differences between TMP and the non-reserve areas sampled, species such as snapper, red moki, parore and butterfish were all more abundant within TMP, demonstrating a degree of consistency with CROP.

Temporal UVC data from past CROP surveys had illustrated a linear decline in the mean abundance of snapper between 2003 and 2008; however, present data indicate high snapper abundance for both sub-legal (< 270 mm) and legal-sized ( $\geq$  270 mm) individuals. A high abundance of sub-legal (0+ and 1+) individuals was also feature of non-reserve areas adjacent CROP, TMP and to a lesser extent non-reserve areas adjacent TMP. This pattern is indicative of high-level recruitment across the outer Hauraki Gulf over recent years. For the last 2-3 years the oceanographic climate in New Zealand has been typified by strong La-Niña conditions (Fig 3.18, NIWA 2011), which for north-eastern New Zealand produces increased wind and swell from the north-east (on-shore) and higher than average sea surface temperature (SST). Consequently, warmer than average SST is favorable for recruitment of species such as snapper (MFish 2011), which is consistent with the data presented here. Traditionally UVC is not the preferred method of estimating snapper abundance as the technique has been implicated in providing unreliable estimates of density, predominantly for legal-sized individuals and as such, BUV is the preferred method of estimation (Willis and Babcock 2000). While we are sympathetic to this view, we feel that if done consistently UVC remains a useful tool, particularly for providing information on snapper recruitment pulses.

Several studies have attempted to discern differences in snapper abundance between reserve and non-reserve areas on the basis of movement behaviour (Egli and Babcock 2004, Parsons *et al.* 2010). Parsons *et al.* (2010) examined concurrent movement behaviour of snapper inside and adjacent to CROP marine reserve indicating that non-reserve snapper had larger home ranges and utilised more than one main area (bi-modal home range), whereas reserve snapper had higher site fidelity with only one main area of use (uni-modal home range). Explanations given for these differences include increased shelter offered by higher abundance of *Ecklonia radiata* inside CROP reserve, and that fish inside CROP reserve may be subject to different rates of fishery induced selection due to their different movement behaviour, i.e., individuals within the centre of the reserve were found to be less likely to cross the reserve boundary, therefore they were less likely to be removed by fishing (see Parsons *et al.* 2010). Higher snapper abundances

within central areas of CROP as detected by UVC (Fig. 3.11 – this study) and BUV (Taylor *et al.* 2005) tend to support those findings. To give further weight to 2011 UVC results, a BUV survey would be recommended in the near future, as when used concurrently, both BUV and UVC techniques arguably provide a more robust estimate of fish abundance and diversity (Willis and Babcock 2001).

In past CROP BUV surveys, variation in blue cod abundance has been suggested to be negatively correlated with higher seas surface temperatures, i.e., numbers declined between 1997 and 2000 when SST were somewhat higher than average, remained stable between 2000 to 2002 when SST was more constant and increasing slightly between 2003 and 2005 when SST was lower than average (Taylor *et al.* 2005). Interestingly, average blue cod numbers have remained at < 1 per BUV drop within and outside CROP over the period 2000-2007 compared to the late 1990s where abundance was generally higher (Haggitt and Mead 2008). UVC data, while variable, tend to support the notation of blue cod abundance being linked to temperature, with a linear decline in abundance from 2003 to 2011. Climate data suggest that average SST for the period 1990-2000 was cooler than for the period 2001-2010 (NIWA climate data, Leigh Marine Laboratory climate data). Should this trend continue then it may have not only implications for blue cod, but also the reef fish assemblages as a whole (see Masuda 2007; Neuheimer *et al.* 2011).

For the present programme, the UVC technique remains useful for making broad comparisons among areas and detecting patterns in fish assemblages through space and time. Because of various biases associated with the technique, Taylor *et al.* (2005) suggest surveys of this nature 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 CROP reef fish programme was incorporated into a recent international review of MPA research methodologies (Addison 2011). While the programme was considered an exceptional example due to its design and duration one criticism, pertinent to all MPA studies evaluated, was that the programme was not based on a concrete set of conservation objectives, i.e., national guidelines objectives and hypotheses which governs MPA research and monitoring in New Zealand. Presently, national conservation objectives directing long-term MPA monitoring are lacking, but this aspect warrants consideration in the immediate future. Data presented here, in tandem with other monitoring studies e.g., Hahei could aid in the development of these conservation objectives.

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 north-Auckland district, the marine environment is likely to be subject to substantial pressures in the future. Moreover, given that sea surface temperatures are likely to rise through time (NIWA 2011) and that temperature

variability can strongly influence reef fish assemblage composition and variability, it will be important to continue monitoring at the current level.

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