Te Whanganui-a-Hei Marine Reserve Benthic and Lobster Monitoring Programme: May-June 2009 Survey

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Te Whanganui-a-Hei Marine Reserve Benthic and Lobster Monitoring Programme: May-June 2009 Survey

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Summary

Biological monitoring of Te Whanganui-a-Hei Marine Reserve and adjacent non-reserve sites was carried out in May-June 2009. The focus of the study was to resurvey permanent 100m^2 quadrats initially surveyed in 2006, and lobster populations that have been routinely surveyed within the Hahei area since 1996.

As for the 2006 survey, biological habitats in 2009 were typical of those found within north-eastern New Zealand coastal areas with higher algal biomasses within the reserve. Generally, mixed algal stands dominated shallow-water reserve sites < 8 m depth, with the laminarian alga *Ecklonia radiata* abundant at depths > 10 m depth. In comparison to 2006 macroalgal assemblages, *Carpophyllum maschalocarpum* has increased in density and biomass within shallow-water reserve sites in 2009 reflecting demographic changes associated with growth, whereas *Ecklonia radiata* biomass has reduced at several deep-water sites due to canopy regression (dieback). Largest changes in assemblage structure were however observed in shallow-water non-reserve sites, whereby fucalean algae has increased in diversity, abundance and biomass within the majority of permanent plots sampled. It is unclear what the main mechanisms driving these changes are, but factors that affect urchin abundance and behaviour may be important. Deep-water non-reserve sites remained relatively unchanged, being dominated by *Ecklonia radiata*. For the most-part, macroalgal assemblage structure across reserve and non-reserve sites showed strong accordance with depth.

In 2009, the abundance of *Evechinus chloroticus* had increased across reserve sites surveyed and decreased across non-reserve sites relative to 2006 levels. Despite this decline urchin numbers remain substantially higher within the non-reserve sample area, although the frequency of urchins displaying cryptic behaviour has increased within all reserve sites and the majority of non-reserve sites sampled. Such behaviour is suggestive of higher predation rates on urchins and/or the effects of higher macroalgal cover.

Levels of fine sediment on reefal habitat has increased within reserve sites in 2009 compared to 2006 levels, illustrating that the Whitianga Harbour continues to have an impact within Te-Whanganui-a-Hei Marine reserve; in contrast the majority of non–reserve sites had low sediment levels. Increased sedimentation between 2006 and 2009 may be reflective of the present day La Niña climatic period. Patterns of this nature are of concern given that sedimentation can have adverse impacts on subtidal marine communities and that the reserve areas that have high sediment levels are biologically diverse.

In 2009, the mean lobster (*Jasus edwardsii*) abundance of 20.3 lobsters per 500m^2 was around five times higher within the reserve compared to unprotected non-reserve sites (4.3 lobsters per 500m^2), and mean lobster size was also significantly higher within the reserve. Temporal data suggest that the reserve population has remained reasonably stable over the last 2-3 years, following a reduction in abundance of legal-sized lobsters within the reserve population between 2004 and 2006. Population stability within the reserve has been the product of high-density recruitment and subsequent on-growth of lobsters into the adult population. Despite higher abundances of lobsters in non-reserve sites in 2009 relative to previous surveys, the low numbers of legal-sized lobsters in the sample population suggest that fishing pressure is continuing to restrict any population growth.

Contents

1.0 Introduction

Monitoring community structure, species diversity, and the distribution and abundance of dominant species 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). With regard to the marine environment, 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), in order to detect processes responsible for determining habitat structure.

No-take marine reserves (marine protected areas) provide a useful tool for monitoring habitat change (Parsons *et al.* 2004) and gauging the response of marine communities and exploited species in relation to protection. Studies in Australasia have provided convincing evidence of the conservation value of no-take Marine Reserves 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, Langlois *et al.* 2005, 2006). However, recovery processes in reserves are complex, and may vary considerably among locations.

In recent years, the Department of Conservation (DoC) has prompted biological monitoring programmes within marine reserves to assess how habitats and biodiversity vary within reserves relative to equivalent unprotected areas (e.g., Shears *et al.* 2000; Shears and Babcock 2002, 2003). This information, in turn, influences the way reserves are managed.

1.1 Te Whanganui-a-Hei (Hahei) Marine Reserve

Te Whanganui-a-Hei Marine Reserve was gazetted in 1993. The marine reserve is situated on the north-facing coast on the Coromandel Peninsula (see Fig. 2.1). The western end of the reserve is flanked by Mercury Bay, which is sheltered and shallow. Several estuaries and rivers flow into Mercury Bay bringing substantial inputs of freshwater and sediment (Shears *et al.* 2000; Schwarz *et al.* 2004). The eastern end of the reserve boarders Mahurangi Island and Hahei Beach and the coast to the east of the reserve is exposed to easterly and southeasterly swells and generally has clearer water. As such, a distinct environmental gradient exists with progressively clearer water from west to east. The reserve encompasses several small nearshore islands and isolated patch reefs. The subtidal rocky reef extends down to a depth of between 3-11 m Mean Low Water Spring (MLWS) along the mainland coast, and extends down to between 20 to 30 m (MLWS) around the islands.

Several biological monitoring programmes have been carried out in the Te Whanganui-a-Hei Marine Reserve since its establishment. Lobster (*Jasus edwardsii*) size and abundance have been monitored since 1996, reef fish since 1997 and benthic communities surveyed in 1999/2000, 2002, 2006 and 2007. In 2001, a sonar side scan of the reserve was produced (Waikato University) and to ground truth this scan, a habitat survey was carried out in 2004 (Hewitt *et al.* 2004). In 2003, the effects of sedimentation on algal community structure including seaweed epifauna along the gradient of the marine reserve (west to east) were studied by NIWA (Schwartz *et al.* 2004) and a study focused on the impact of sedimentation on invertebrate encrusting communities was undertaken between 2003 and 2006 (Steger 2007).

The first thorough biological monitoring study and habitat description of Hahei was conducted by Shears *et al.* (2000) for DoC. In that study, four sites within the marine reserve and five sites outside of the marine reserve were surveyed using depth stratified sampling. Reef communities were found to vary significantly between sites with the variation accredited to changes in wave exposure and sedimentation from sheltered turbid sites within Mercury Bay to the exposed island sites which experience clearer offshore waters. Reserve related effects were evident within the 4-6 m depth strata with significant differences between reserve and non-reserve sites, with higher density and biomass of large brown macroalgae in the reserve and a higher abundance of the urchin *Evechinus chloroticus* in non-reserve areas. Urchin population structure also differed at reserve sites, but due to the high natural variability among sites, reserve-related effects were difficult to discern.

Schwarz *et al.* (2004) assessed potential direct and indirect effects of terrestrial runoff on rocky reef biodiversity between Whitianga River mouth and Hahei Beach (from west to east), which encompasses the gradient in water turbidity. The study found no clear difference in understorey species across this gradient, but suggested reduced productivity of *Ecklonia radiata* at the most consistently turbid site, Cooks Bluff.

In addition to these biological monitoring programmes a range of social studies have been carried out in the marine reserve and include an assessment of the social-economic effects of the establishment of the marine reserve in 1995, 1997 and 2002, a study of community attitudes towards the marine reserve in 1994, a 10 year assessment of the marine reserve in 2002 (looking at impacts, knowledge, opinion and use), a recreational reserve visitors survey in 1998 and to quantification of human activities in and around the marine reserve in 2006. Human activities in and around the marine reserve have the potential to impact on the reserve biota and information about these activities is necessary to be able to take well-informed management decisions.

1.2 Present study

The Waikato Conservancy of DoC require biological monitoring of the Te Whanganui-a-Hei Marine Reserve in 2009 surveying permanent quadrat first established and surveyed in 2006. The components of the study include benthic and lobster monitoring with a third component, reef fish monitoring, to be undertaken in 2010.

Monitoring programs carried out in the Te Whanganui-a-Hei Marine Reserve from 1993 to 2004 have had a strong focus on determining the effects of marine reserve protection. This aim is still pertinent, but since the effectiveness of the marine reserve has been proven over the last 12 years, the main objective has now changed to long term trend detection that is capable of identifying threats to the reserve biota over which DoC can exert some management control, or which have the potential to cause catastrophic changes to particular species and/or sections of the marine community.

The overall objectives of the programme are to:

- Determine natural levels of temporal and spatial variation in reserve biota and habitats;
- Assess biodiversity within reserve and non-reserve areas;
- Detect changes that differ significantly from natural variation;

- Detect introduced species that may have a direct or indirect effect on the natural functioning of reserve ecosystems;
- Determine the current population status of lobster (*Jasus edwardsii*) within and between reserve and non-reserve areas;
- Compare lobster size and abundance within and between reserve and non-reserve areas;
- Compare trends in Te Whanganui-a-Hei Marine Reserve lobster populations through time;
- Link into regional, national and, if possible, international marine reserve monitoring programmes;
- Assist with research on marine ecosystem and marine reserve functioning.

2.0 Methodology

The biological monitoring programme (including lobster surveys) for this report focuses on rocky reef sampling within reserve and non-reserve locations. Site selection was based on a marine reserve habitat map constructed by NIWA in 2004 (herein NIWA 2004; also see Hewitt *et al*. 2004). The map includes three rocky reef habitat types within the reserve, as well as several areas of special interest within and outside of the reserve (Table 2.1; see Appendix 3).

Table 2.1 Three habitat types used to construct the Te Whanganui-a-Hei Marine Reserve map (NIWA 2004).

Habitat	**Description**
Rocky reef: Habitat Type 1	*Ecklonia* forest, foliose algae with some sponges, turfing algae and sand
Rocky reef: Habitat Type 2	Sponge flats, foliose algae, turfing algae and sand
Rocky reef: Habitat Type 3	Algae and sand and some sponge flats

2.1 Rocky reef sites

A total of 10 rocky reef sites within the Hahei Marine Reserve and 10 rocky reef sites outside of the marine reserve were sampled between 5 May and 2 June 2009 (Fig. 2.1). Sites within the reserve comprised four sites used previously for lobster surveys (see Haggitt and Kelly 2004) and an additional six sites randomly selected from within the three rocky reef habitat types (Table 2.1) in 2006 (Appendix 3).

Similarly, for non-reserve sites, four sites previously used for lobster surveys were sampled along with an additional six rocky reef sites from random selected localities in 2006 (Appendix 3). Initially, these sites were to be selected from a total of thirty additional sites. However, as many of those sites were found to occur on soft-sediment habitat, rocky reef sites were selected from *ad hoc* dives and video drops (Haggitt and Mead 2006).

Permanent sites

At each site, one permanent 100 m^2 quadrat was sampled to quantify species abundance and percent cover. This approach has been used previously within other marine reserves e.g., Cape Rodney to Okakari Point Marine Reserve (Shears and Babcock 2003) and other coastal monitoring studies (e.g., Anderson *et al*. 2005). For relocation of permanent sites within rocky reef habitats, each quadrat was delineated with stainless steel markers and subsurface buoys, fixed to the substratum with Expocrete® cement.

Sampling followed the general methodologies of Shears and Babcock (2003) for quadrat sampling and within each 100 m^2 permanent plot, ten haphazardly placed 1 m^2 quadrats were sampled by counting and measuring all large brown macroalgae and invertebrate taxa within (see below).

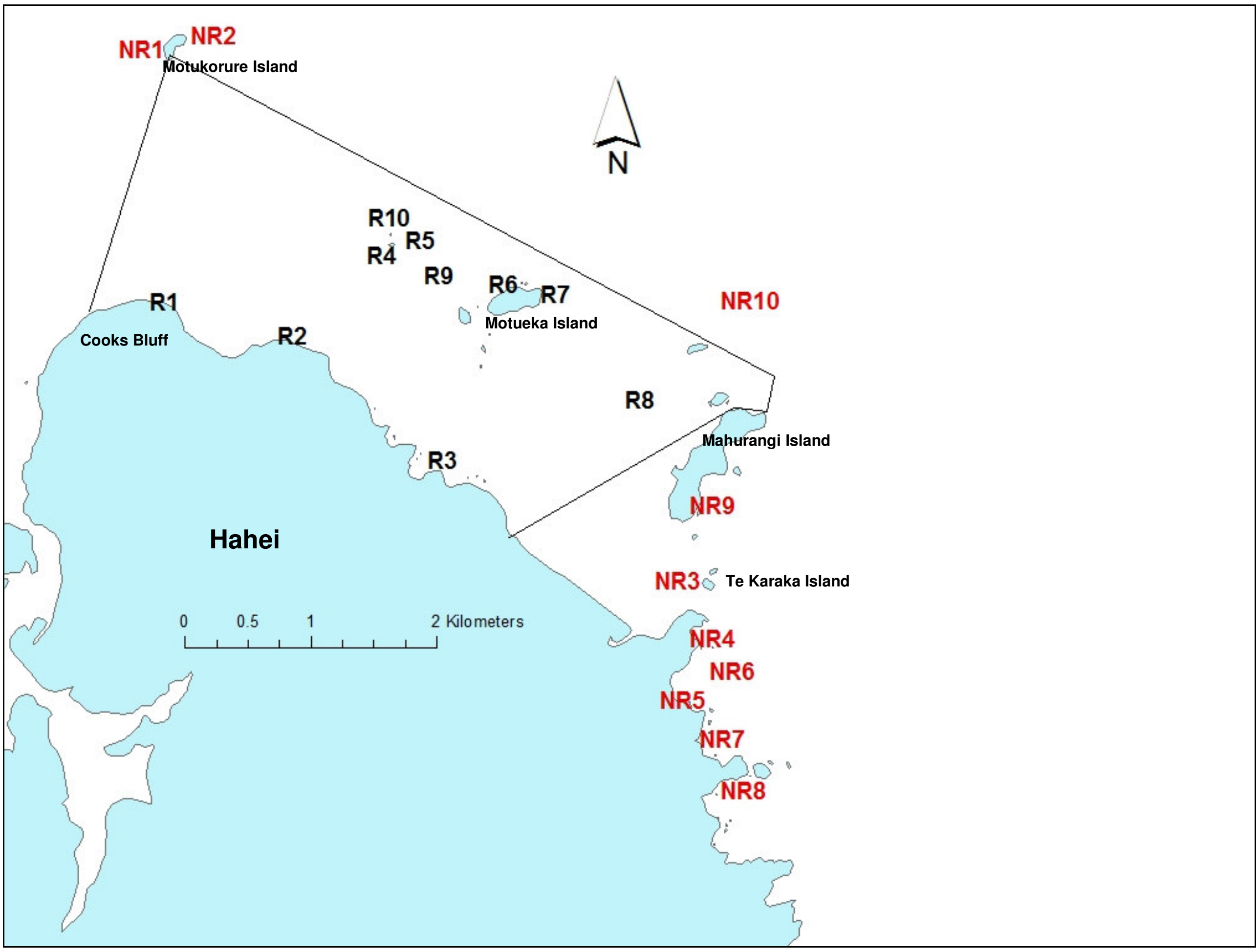

Figure 2.1 Location of sampling sites within Te Whanganui-a-Hei (Hahei) Marine Reserve (depicted by dark outline) and non-reserve areas in 2009. Reserve sites are denoted in black R1 to R10, and non-reserve sites in red (NR1-NR10). Refer to Appendix 3 for site coordinates and average depths for each site. R1 (Cook Bluff shallow); R2 (Boulder Bank shallow); R3 (Gemstone north shallow); R4 (Moturoa Island deep); R5 (Moturoa north-east deep); R6 (Motueka Island shallow); R7 (Motueka Island deep); R8 (Kingfish Reef deep); R9 (Motueka Island north-west deep); R10 (Moturoa Island north-west deep); NR1 (Motukorure Island shallow); NR2 (Motukorure Island deep); NR3 (Te Karaka Island); NR4 (Hereheretaura Point shallow); NR5 (Whitecliffs shallow); NR6 (Whitecliffs deep); NR7 (Te Puphua Point north shallow); NR8 (Te Puphua Point south shallow); NR9 (Cave Bay deep); NR10 (South Sunk Rock deep).

Macroalgae

All large brown macroalgae and turfing algal species within each quadrat were counted, measured and their percent cover estimated. The total length of all brown algae was measured to ± 5 cm and individual measurements of stipe length and primary lamina length were made to ± 5 cm for the laminarian algae *Ecklonia radiata* and *Lessonia variegata*. Macroalgal length measurements were then converted to biomass based on length-dry weight relationships presented in Shears and Babcock (2003) (see Table 2.2).

Table 2.2. Algal species and functional groups used in analysis along with length-weight and/or percent cover-weight relationships for biomass estimates. y = dry weight (g), x = total length (cm), SL = stipe length (cm) and LL = laminae length (cm). Data are from Shears and Babcock (2003).

Brown algae	
Carpophyllum angustifolium	$y = 0.068x - 0.27$
C. maschalocarpum	$\ln(y) = 1.764\ln(x) - 4.311$
C. plumosum	$\ln(y) = 1.472\ln(x) - 3.850$
C. flexuosum	$\ln(y) = 2.049\ln(x) - 5.251$
Xiphophora chondrophylla	$y = 1.786x - 4.171$
Ecklonia radiata – Stipe	$\ln(y) = 1.671\ln(\mathrm{SL}) - 3.787$
– Laminae	$\ln(y) = 1.177\ln(\mathrm{SL} \times \mathrm{LL}) - 3.879$
Sargassum sinclairii	$y = 0.075x + 0.124$
Landsburgia quercifolia	$\ln(y) = 1.971\ln(x) - 5.058$
Small brown algae,	$\ln(y) = 2.587\ln(x) - 6.443$
e.g. *Zonaria turneriana*	1% = 2.5 g
Brown turf, e.g. *Distromium*, *Dictyota* spp.	1% = 1.5 g
Brown encrusting, e.g. *Ralfsia*	1% = 0.1 g
Red algae	
Osmundaria colensoi	$\ln(y) = 1.720\ln(x) - 3.379$, 1% = 22.9 g
Pterocladia lucida	$\ln(y) = 1.963\ln(x) - 5.076$ 0., 1% = 10.0 g
Melanthalia abscissa	$\ln(y) = 1.775\ln(x) - 4.247$
Red foliose, e.g. *Plocamium* spp.	$\ln(y) = 2.649\ln(x) - 8.812$
Red turfing (< 5 cm), e.g. *Champia* spp.	1% = 1.7 g
Coralline turf, e.g. *Corallina officinalis*	1% = 4.5 g
Crustose corallines	1% = 0.1 g
Red encrusting	1% = 0.1 g
Green algae	
Codium convolutum	1% = 4.7 g
Others, e.g. *Ulva* sp.	1% = 1.7 g
Filamentous algae	1% = 0.2 g

Encrusting species

The primary (substratum) percent cover of foliose algae, turfing algae, encrusting algal species, encrusting invertebrates (e.g., sponges and ascidians bryozoans) as well as sediment and sand cover were recorded in each 1m^2 quadrat using a visual estimation technique (see Shears and Babcock 2003). Briefly, quadrats were divided into quarters (1/4 =25 %) to assist in estimating covers of dominant forms, while the covers of minor forms were estimated on the basis that a 10 x 10 cm area equates to 1 % cover. This technique is considered to be the most suitable for this study as it is efficient and ensures that the cover of all forms are recorded, unlike point-intercept methods. Sub-samples of any unidentifiable species were taken, preserved, and then identified at the Leigh Marine Laboratory.

Urchins

All urchins occurring within each 1m^2 quadrat were counted and their behavioural characteristics noted, i.e., grazing in the open (exposed behaviour) or in crevices and holes (cryptic behaviour). The test diameter of all urchins > 10 mm was measured to the nearest 5 mm and an assessment of urchin health also made. Three health categories were used to grade urchin health: 1=Healthy (all spines intact); 2=Loss of guard spines; 3=Complete loss of all spines with bald areas of test visible.

Gastropods

All gastropods on the substratum and on macroalgae within each 1m^2 quadrat were counted and the largest shell dimension (width or length) for each species measured. For example, shell width was measured for *Cookia sulcata*, whereas shell height was measured for *Cantharidus purpureus*. The total length of paua (*Haliotis* species), limpets (*Cellana stellifera*) and chitons was also measured.

Note: All animal taxa enumerated in the survey were checked using the New Zealand Inventory of Biodiversity (Gordon 2009).

Invasive species

In recent years, parts of the New Zealand coastline have been subject to several invasive species introductions such as the laminarian *Undaria pinnatifida,* the 'solitary' sea squirt (clubbed tunicate) *Styela clava*, and the paddle-crab *Charybdis japonica*. All permanent quadrats and areas adjacent to each quadrat were checked for the possible occurrence of these taxa and other invasive taxa such as the green alga *Caulerpa taxifolia* that could be potential threats to the New Zealand coastline and biodiversity.

Environmental variables

A range of physical variables: rock type, wave exposure, distance from Whitianga Harbour, depth and sediment percent cover were assessed for each site as part of the study. Physical variables were used to assess the variability in biological datasets.

Rock type

The nature of the rock type within each permanent quadrat was recorded based on 5 categories:

- Low lying platform reef;
- Boulder reef;
- Platform and boulder reef mix;
- Cobbles;
- Complex platform reef characterised by overhangs and crevices.

Oceanographic climate

Data from MarineWeather.co.nz (MW) were used examine the oceanographic climate within the Hahei region between 2006 and 2009. MW is a forecast model that predicts shallow-water wave conditions at ~500 locations in New Zealand. MW uses NOAA WaveWatchIII deepwater wave solutions as input along with detailed shallow-water bathymetry data and the SWAN (Simulating Waves Nearshore) model. MW has very good spatial coverage however; it has only been running since 2006, so long time-records are not available. Regardless, archived wave data for Hot Water Beach were extracted to gauge the recent wave climate for this region of New Zealand.

Wave exposure

Using data from MW (above) wave exposure for each of the survey sites was calculated using WBFORM within 3DD suite (ASR Ltd). This method provides a normalised wave height for each site, which was then used as a surrogate for wave exposure.

Distance from Whitianga Harbour

The influence of Whitianga Harbour has been attributed to influencing biological populations within Te Whanganui-a-Hei marine reserve (Steger 2006). To provide a measure of this, the distance (± 0.05 km) of each site from Whitianga Harbour mouth was determined using the measure tool in ArcMap as a variable to be used in further analysis (see data analysis below).

Video data

In addition to quantitative monitoring, a video drop was done within each sampling site to provide additional information on the distribution of dominant organisms. Sampling was carried at each site using a Splashcam® underwater camera unit connected to Sony® digital videocassette recorder (GV-D800E) on the surface. Each site was videoed for a minimum of 5 minutes. Following video sampling, the digital video was then transcribed to computer and hyperlinked to a location map of the region.

Data analysis

Unless otherwise stated, means are given ± their associated standard error (SE). A combination of multivariate and univariate statistical tests was used to analyse benthic rocky reef data for 2006 and 2009 surveys.

Multivariate

To test for overall differences in algal communities (based on biomass values) between reserve and non-reserve sites, non-parametric multivariate analysis of variance (PERMANOVA) (Anderson 2001) was used. The multivariate null hypothesis was that there is no difference between reserve and non-reserve areas between surveys. Non-parametric tests based on permutation such as PERMANOVA are generally favoured over traditional parametric MANOVA, because ecological data very rarely conforms to the strict assumptions of these tests (e.g., normality). Analysis was carried out on biomass data for the most dominant algal species (5 taxa - *Ecklonia radiata, Carpophyllum flexuosum, Carpophyllum maschalocarpum, Carpophyllum plumosum,* and *Xiphophora condrophylla*); while all macroalgal taxa (21 in total) were analysed separately (refer to Table 2.2). Main factors in the analysis were ‘Status’ (reserve, non-reserve) and ‘Year’ (2006 and 2009 surveys), which were treated as fixed factors, whereas the factor the factor 'Site' nested within Status and Year, i.e., 'Site(Status×Year)' was treated as a random effect.

To further visualise macroalgal biomass patterns in multivariate space, non-metric MDS analysis was also undertaken for the factor Status and Depth using Primer-E (Clark and Warwick 2001). The importance of 5 environmental factors (Depth (m), Wave Exposure (normalized long-term wave exposure (0-1)), Sediment level (% Cover), Substratum Type (see above), and Distance from the Whitianga Catchment (km)) in explaining the variation in macroalgal and sessile invertebrate assemblages was tested using multivariate multiple regression (DISTLM v2; Anderson 2002). The multivariate null hypothesis was that there is no relationship between macroalgal communities between reserve and non-reserve areas and environmental variables.

All multivariate analyses were based on Bray-Curtis similarities, with all data fourth-root transformed prior to analysis. The same multivariate techniques used to analyse macroalgal data were also applied to sessile invertebrate data for 2006 and 2009 survey data.

Biodiversity indices were calculated for macroalgae, mobile invertebrates, and sessile invertebrates using the routine 'Diverse' in Primer-E. Key indices calculated were taxa richness, Margalef diversity, Shannon-Weaver diversity, and Simpson's diversity.

Table 2.3. Indices used to describe macroalgal, sessile invertebrate, and mobile invertebrate biodiversity.

Biodiversity index	Notes
Taxa richness (S)	Total number of species in a given sample or area.
Margalef (d)	Richness index standardizes the number of *N* ln species encountered against the total number of individuals encountered.
Shannon-Weaver (H')	Used to measure diversity for categorical data. The Shannon-Weaver diversity index is based on the number of different species per sample (species richness) and the 'relative abundance' of the different organisms present.
Simpson 's (1-λ')	Measures the probability that two individuals randomly selected from a sample will belong to the same species (or some category other than species).

Univariate

Univariate statistical tests were used to complement multivariate statistical tests and examine differences in count data for individual species between reserve and non-reserve areas for 2006 and 2009 surveys. The univariate null hypothesis was that there is no relationship difference between the response variable (e.g., Urchin density) between reserve and non-reserve areas and between surveys. Prior to analysis, data were checked for normality of errors and homogeneity of variances. Because much of the data violated assumptions of traditional linear models such as ANOVA[1], generalised linear mixed modeling using the SAS macro GLIMMIX (Littell *et al.* 1996, SAS 1999) was employed. For the purposes of the test, the factors 'Status' and 'Year' were treated as fixed factors whereas the factor 'Site' nested within Status and Year 'Site (Status×Year)' was a random effect in the model (see Zar 1999). For all count data, the model was back-fitted to a Poisson distribution using a log-link

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[1] Residual plots of count data showed numerous outliers whereas Shapiro-Wilk *W*-tests for normality of errors and Levene's test for homogeneity of variances were significant in many cases. Appropriate transformations (Zar 1999) failed to ameliorate these problems.

function and the dispersion parameter Scale=Deviance was employed to account for any overdispersion in the dataset.

To test for differences between urchin size data between reserve and non-reserve areas and between surveys a Two-Way ANOVA (SAS 1999) was employed.

2.2 Lobster

Jasus edwardsii size and abundance was assessed at each site by sampling three 50 m x 10 m haphazardly placed transects. The size, and where possible, sex of all lobsters within each transect was determined by visual estimation (see MacDiarmid 1991). The choice of the 50 m x 10 m transect size was based on a pilot study conducted by MacDiarmid (1991) who compared the precision of 3 different transect sizes, 10 m x 10 m (n=20), 25 m x 10 m (n=8) and 50 m x 10 m (n=4), each covering a total area of 2000 m^2. MacDiarmid (1991) found that all transects provided a similar level of precision. The 50 m x 10 m transect was selected for the Hahei Marine Reserve survey to permit at least one transect per dive to be completed in areas of high lobster abundance, and to limit the number of zero counts in areas of low lobster abundance. The number of sites surveyed in 2006 was increased from four sites to ten sites, but the level of replication for this survey was reduced from that used in previous surveys from 5 to 3 transects. The same survey method used in 2006 was employed for the 2009 survey.

Sex was determined using the dimorphic characteristics of male and female lobsters. Torches were used to aid in the sexing of lobsters and to ensure that lobsters in deep holes were not missed. All divers were required to estimate carapace length to within an average of 10 mm. This level of accuracy was achieved through a series of calibration dives where the size of individual lobsters was estimated, after which each lobster was caught by hand and measured with vernier calipers to obtain a true length measurement (Fig. 2.2). An analysis of covariance (ANCOVA) could not detect any significant difference between the size estimation ability of the three censors used in the survey, i.e., the slope was not significantly different from 1 ($P = 0.595$) and the y intercept did not differ significantly from 0. In northern New Zealand, the minimum legal size limit for *J. edwardsii* occurs between 95 mm and 100 mm C.L. For the purpose of this report lobsters ≥ 95 mm were therefore considered to be legal and thus susceptible to fishing.

Data analysis

Abundance and size data is presented graphically. Prior to formal analysis, data were tested for normality and homogeneity of variances with a Shapiro-Wilk *W* test and residual plots. As the 2009 data violated assumptions of traditional ANOVA (as above), generalised linear mixed modeling GLIMMIX (as above) was used to analyse 2009 abundance data and temporal data 2006 to 2009. Again, the factors 'Status' and 'Year' were treated as fixed factors whereas the factor 'Site' nested within Status and Year 'Site (Status×Year)' was a random effect in the model. For lobster data, ratios of density (plus 95% confidence limits) were calculated between significant levels to provide an estimate of the size of main effects. *Note*: confidence limits are asymmetrical as they are calculated on the log-scale.

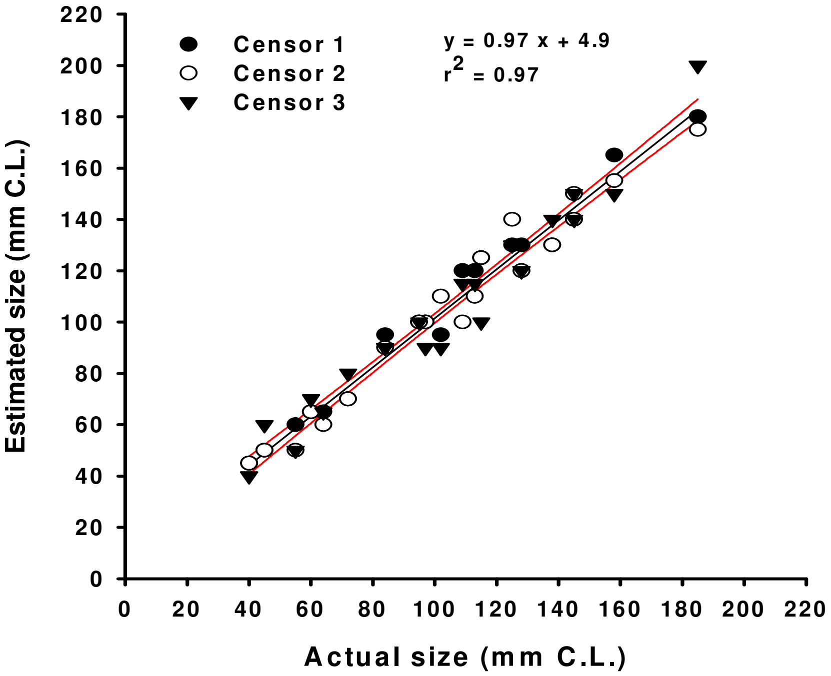

Figure 2.2. Size calibration data from the three censors conducting the 2006 survey of Hahei Marine Reserve and adjacent coastline. Size estimates were made without handling individual lobsters. Actual sizes were determined by capturing the lobsters and measuring with vernier calipers after the size estimates were made. The least squares regression line for the pooled estimates (± 95% confidence intervals *in red*) is also given.

3.0 Results

Results from the 2009 survey are presented for macroalgae, sessile invertebrates, mobile invertebrates and lobster. Reference is made throughout to the initial 2006 survey data, which if not presented either graphically or in table format in the main body of the results section, are presented sequentially in Appendix 1. Full species lists are also presented in Appendix 2.

An interactive CD-Rom accompanies this report (back cover). It includes video data from each survey site and a GIS module that contains biodiversity indices (taxa richness (S), Margalef richness (d) Shannon-Weaver (H'), and Simpson's diversity ($1-\lambda'$)) for macroalgae, mobile invertebrates and sessile invertebrates. For conciseness, only taxa richness is graphically presented in the main body of the results section.

3.1 Macroalgae

Ecklonia radiata

Analogous to the 2006 survey, the stipitate laminarian alga *Ecklonia radiata* was the most ubiquitous algal species sampled within the reserve. *Ecklonia radiata* occurred at all reserve sites generally maintaining highest densities and biomasses in deep-water (Fig. 3.1), the latter due to a higher frequency of larger sporophytes (SL > 700 mm) that typically formed enclosed canopies with juvenile stages (SL < 100 mm) also common (Fig's 3.1, 3.2). Reserve sites dominated by *Ecklonia radiata* included R2 (Boulder Bank Shallow), R4 (Moturoa north-east deep), R5 (Moturoa Island deep), R7 (Motueka Island deep), R8 (Kingfish Reef deep), R9 (between Motueka Island and Poikeke Island), and R10 (north-west Moturoa Island north-west deep). Mean densities ranged between 7-10 sporophytes m^{-2} across these sites (Fig. 3.1). While dense *E. radiata* stands were also apparent at the shallow-water site R2, *E. radiata* was generally patchy distributed at the other shallow-water sites sampled (R1, R3 and R6) co-occurring with fucalean algae such as *Carpophyllum maschalocarpum* and *Carpophyllum flexuosum* (Fig. 3.4).

Biomasses of *Ecklonia radiata* in 2009 were generally lower than 2006 levels for deep-water sites R4 and R9, but higher for site R7 and the shallow-water site R2 (Fig. 3.1). Lower biomasses at R4 and R9 in 2009 may be related to the condition of the stands sampled which were exhibiting signs of canopy thinning and patchy dieback synonymous with canopy degeneration (Cole and Babcock 1996, Haggitt and Babcock 2003). Based on long-term demography studies of *Ecklonia radiata* stands at depth, the dieback and associated reduced biomass to be a temporary event and the occurrence of juvenile sporophytes on the immediate substratum, particularly at R4 suggests that habitat structure is unlikely to alter drastically in the long-term.

Ecklonia radiata also occurred at all non-reserve sites sampled in 2009 (Fig. 3.1), but only formed enclosed canopies at the deep-water sites NR2 (Motukorure Island deep), NR6 (Whitecliffs deep) and NR9 (Cave Bay deep – Mahurangi Island). Across these sites, densities ranged from 10-15 sporophytes per m^2 with biomasses around 200 g dwt^{-1} m^{-2}; equivalent to many of the deep-water reserve sites (Fig. 3.1). Patchy canopy regression (dieback) was also was observed at NR2 (Fig. 3.3). Low density, patchily distributed, *Ecklonia radiata* stands were characteristic of all non-reserve shallow-water sites surveyed, with larger sporophytes commonly < 700 mm SL (Fig. 3.2). Interestingly, in 2006, *Ecklonia radiata* was absent from all non-reserve shallow-water permanent quadrats.

The most obvious difference between reserve and non-reserve areas surveyed in 2006 was the general lack of *Ecklonia radiata* stands in non-reserve shallow-water permanent plots. The increased density and corresponding biomass of *Ecklonia radiata* at these sites in 2009 suggests that an element of habitat change has occurred at these sites over this period. Determining whether these observed changes will lead to a more pronounced habitat shift over the longer term at these sites would be of interest.

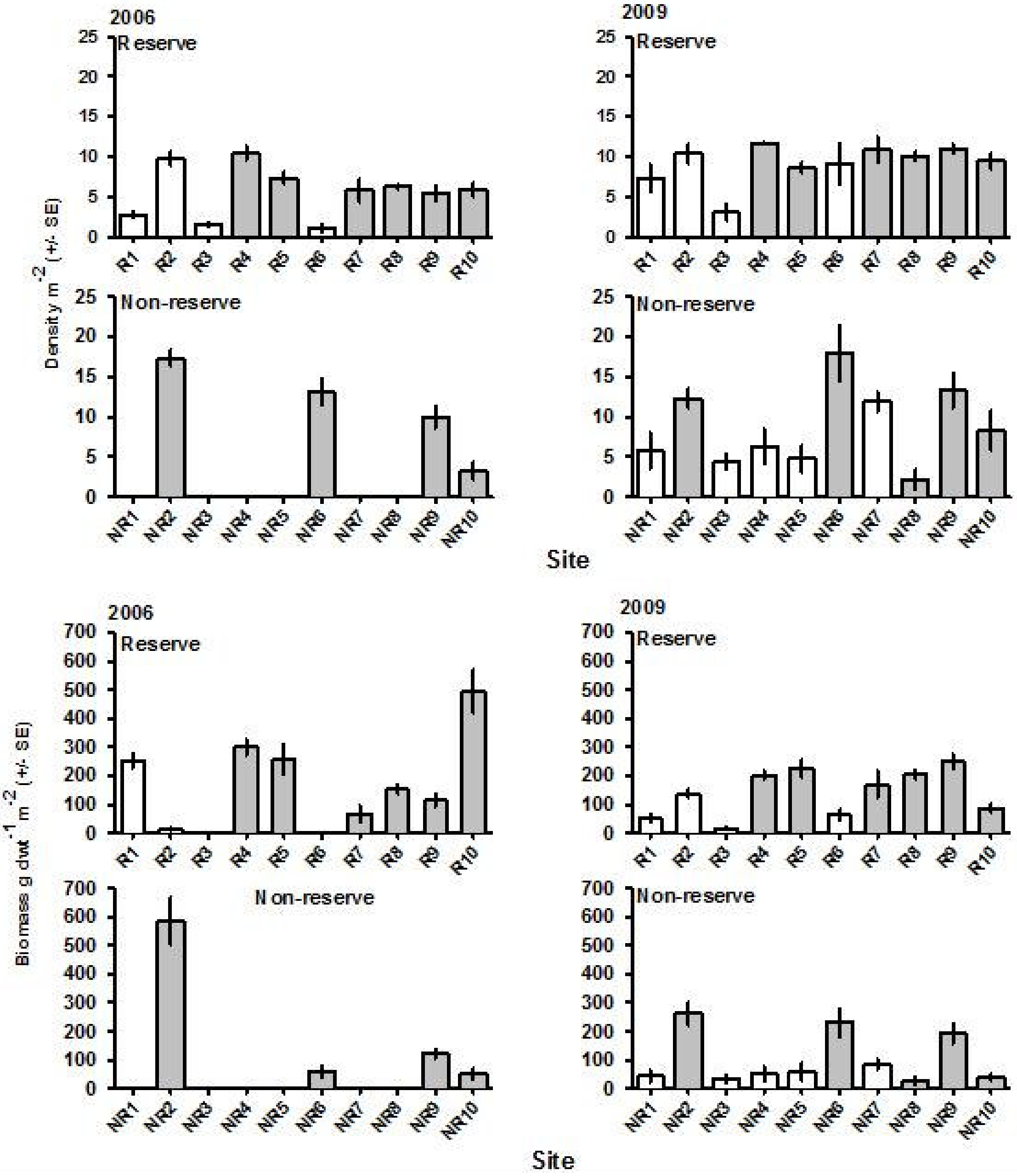

Figure 3.1. *Ecklonia radiata* sample population density and biomass within Te Whanganui-a-Hei Marine Reserve and non-reserve sampling sites in 2006 and 2009. Open bars denote shallow-water sites (< 8 m depth) whereas shaded bars denote deep-water sites (> 10 m depth). Refer to Fig 2.1 for sampling site location.

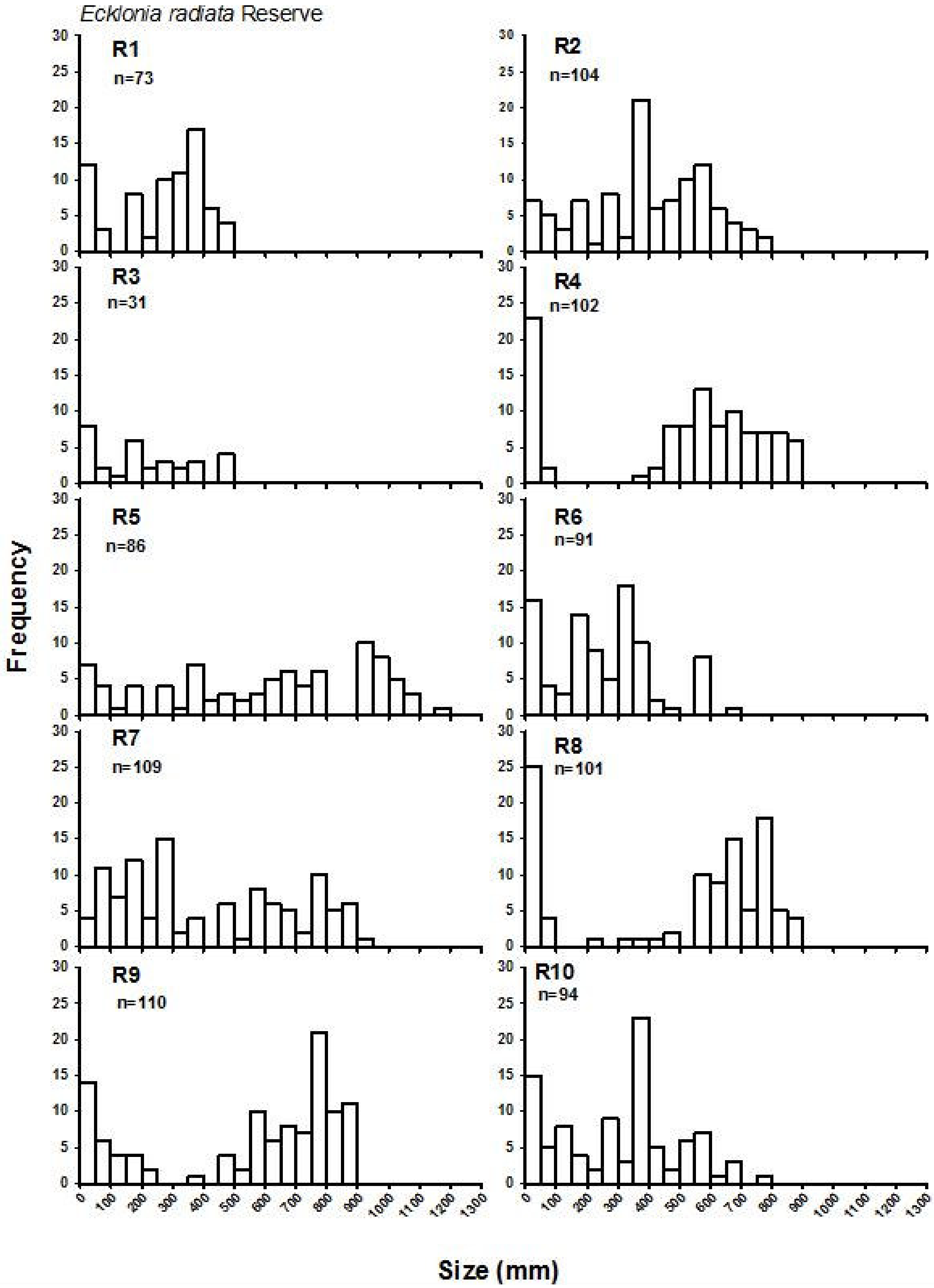

Figure 3.2. *Ecklonia radiata* size frequency distribution (based on stipe length) within Te Whanganui-a-Hei Marine Reserve sampling sites in 2009. Refer to Fig. 2.1 for sampling site location.

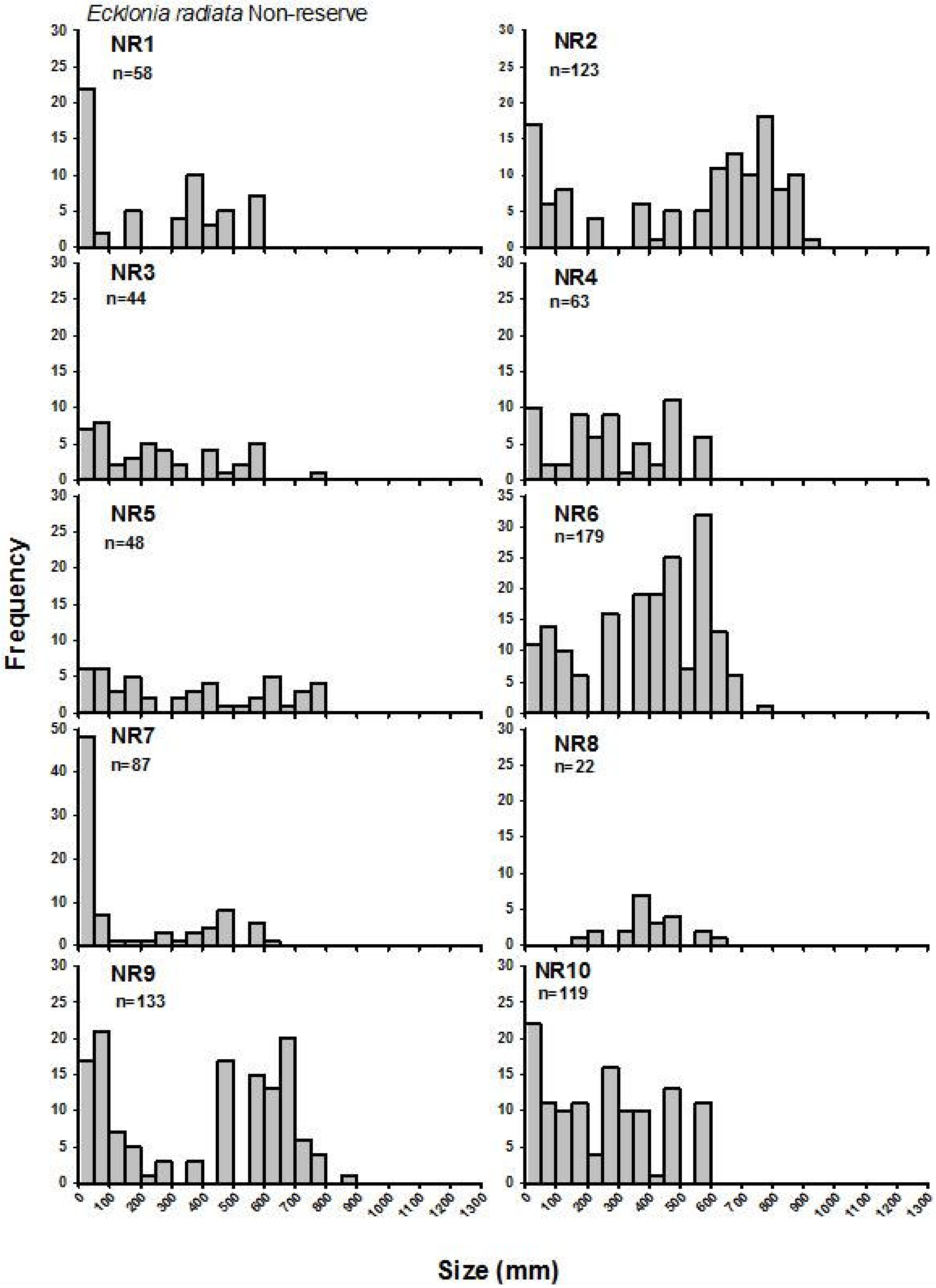

Figure 3.2 continued. *Ecklonia radiata* size frequency distribution (based on stipe length) within non-reserve sampling sites in 2009. Refer to Fig. 2.1 for sampling site location.

Statistical analysis of *Ecklonia radiata* biomass data for 2006 and 2009 surveys using generalised linear mixed modelling (GLIMMIX – SAS 1999) indicated that the factor Status was statistically significant, due to *Ecklonia radiata* biomass being consistently higher within the reserve sample population, yet despite an increase in biomass in shallow-water non-reserve sites the factor Year was not statistically significant. The random effect in the model Site(Status×Year) was also statistically significant, indicative of the high variability in *E. radiata* biomass among sites between surveys (Table 3.1).

Figure 3.3. Patchy *Ecklonia radiata* canopy regression in 2009 at site N2 (adjacent Motukorure Island). This type of canopy regression was also observed at reserve sites R4 and R9 in 2009.

Table 3.1. Results from mixed model analysis (GLIMMIX) for *Ecklonia radiata* biomass from permanent quadrat sampling at reserve and non-reserve sites in 2006 and 2009. Significance: *p < 0.05, **p < 0.01, ***p < 0.001.

Factor	**Fixed effect**			**Covariance parameter estimate**
	Status	**Year**	**Status×Year**	**Site (Status×Year)**
Ecklonia radiata	$F_{1,16} = 2.35$*	$F_{1,16} = 0.96$	$F_{1,16} = 1.34$	Z value = 4.10, Pr $Z < 0.0001$

Fucalean algae

Fucalean algae remained a dominant component of the shallow-water reserve sites surveyed in 2009 and taxa occurrence was broadly consistent with the 2006 survey (Appendix One), several taxa increased in density and biomass between surveys. Of these, *Carpophyllum maschalocarpum* formed dense canopies at sites R1, R2, R3 and the deepwater site R7 (Fig 3.3). *C. maschalocarpum* also had the highest biomass of all fucalean taxa surveyed across sites, exceeding 1000 g dwt^{-1} m^{-2} at R1 and 600 g dwt^{-1} m^{-2} at both R2 and R3 (Fig. 3.4). As recruit stages (< 100m TL) were abundant at R1, R2 and R3 in 2006 (Appendix 1), 2009 size structures reflect the on-growth of these recruitments (Fig. 3.4).

Other fucalean taxa that had increased in density and biomasses between 2006 and 2009 were *Carpophyllum plumosum* at sites R1, R2, and in particular, site R3, and *Xiphophora condrophylla* at sites R1, R3, but has remained stable at R6 (Fig. 3.4). As for the 2006 survey, *Carpophyllum flexuosum* occurred in discrete high-density patches in shallow water at R6 (Shallow Motueka Island), and R3 (Fig. 3.3). For the most-part, fucalean algae remained absent, or occurred at low densities (and biomasses) within deep-water sites, matching depth-related patterns in macroalgal habitat structure within north-eastern New Zealand (Shears *et al*. 2004).

Changes in the abundance and biomass of fucalean taxa between 2006 and 2009 were not only restricted to reserve sites. With the exception of deep-water sites NR2, NR9, and NR10, all non-reserve sites had notable changes in macroalgal assemblage structure based on fucalean diversity, density, size structures and corresponding biomasses (Fig's 3.4, 3.5). In 2006, urchin-grazed barrens habitat dominated sites NR1, NR3, NR4, NR5 and NR8, whereas in 2009, while barrens habitat was still evident within and adjacent to many of the permanent plots, patchy mixed stands of fucalean algae were often predominant. Most-notable changes were at sites NR1, NR3, NR4 and NR5. At NR1 *Carpophyllum maschalocarpum*, *Carpophyllum flexuosum* and *Xiphophora condrophylla* were conspicuous, at NR3 *Carpophyllum flexuosum* and *Xiphophora condrophylla* were dominant, at NR4 *Carpophyllum flexuosum* had increased noticeably, and at NR5 *Carpophyllum maschalocarpum* and *Carpophyllum flexuosum* were prominent. Size frequency plots (Fig. 3.4) suggest that many of the mixed stands are likely to be the product of recent recruitment events, particularly when compared to mixed algal stands within the reserve that have dense and enclosed canopies.

Analysis of temporal biomass data for *Carpophyllum maschalocarpum*, *Carpophyllum flexuosum, Carpophyllum plumosum* and *Xiphophora condrophylla* between surveys (GLIMMIX – SAS 1999) indicated that the factor Status and Year was statistically significant for *Carpophyllum maschalocarpum*, being consistently higher within the reserve sample population and increasing between surveys. The factors Year and Site and the Year×Site interaction were not statistically significant for the other taxa despite increased biomasses in shallow-water non-reserve sites. The random effect in the models Site(Status×Year) were statistically significant for all taxa, reflecting the high variability in biomass among sites between surveys (Table 3.2).

Overall, reserve areas surveyed in 2009 had a higher biomass of large brown algae compared to non-reserve areas and similar to the 2006 survey. In 2006, this difference was primarily due to the lack of urchin-grazed barrens habitat within the reserve, however, this difference may become less significant should *Ecklonia radiata* and fucalean algae become further established at non-reserve sites. Multivariate analysis based on biomasses of the main large habitat forming brown macroalgae using PERMANOVA (*Ecklonia radiata*, *Carpophyllum flexuosum*, *C. maschalocarpum*, *C. plumosum* and *Xiphophora chondrophylla*) indicated that the factor Status was statistically significant (Table 3.3), due to higher overall biomass occurring within the reserve, but the factor Year was not statistically significant. The random factor Site(Status×Year) was highly significant, indicating that biomass at a site-specific level changed at different rates between reserve and non-reserve areas surveyed between 2006 and 2009.

Table 3.2. Results of generalized linear mixed modeling for fucalean algal biomass and crustose coralline algae (CCA) from permanent quadrat sampling at reserve and non-reserve sites in 2006 and 2009. Significance: *p < 0.05, **p < 0.01, ***p < 0.001. Models back-fitted by removing non-significant interaction terms.

Fixed effects				**Covariance parameter estimates**
Factor	Status	Year	Status× Year	Site (Status × Year × Depth)
Carpophyllum flexuosum	$F_{1,16} = 0.134$	$F_{1,16} = 0.53$	$F_{1,16} = 0.38$	Z value = 2.73, Pr $Z < 0.001$
Carpophyllum maschalocarpum	$F_{1,16} = 4.49$*	$F_{1,16} = 4.55$*	$F_{1,16} = 2.12$	Z value = 2.38, Pr $Z < 0.05$
Carpophyllum plumosum	$F_{1,16} = 0.53$	$F_{1,16} = 3.10$	$F_{1,16} = 0.65$	Z value = 4.12, Pr $Z < 0.0001$
Xiphophora condrophylla	$F_{1,16} = 2.32$	$F_{1,16} = 0.63$	$F_{1,16} = 0.27$	Z value = 5.45, Pr $Z < 0.0001$
CCA	$F_{1,16} = 5.67$*	$F_{1,16} = 4.15$	$F_{1,16} = 0.49$	Z value = 2.55, Pr $Z < 0.0001$

Table 3.3 Results from PERMANOVA (degrees of freedom, *F* ratios and *P* values from permutation) for biomass of five dominant brown macroalgae from permanent quadrat sampling at reserve and non-reserve sites between 2006 and 2009 surveys.

Source	**df**	***F***	***P* (perm)**
Year	1	2.2254	0.0843
Status	1	5.2325	0.0022
Site(Year×Status)	36	19.4779	0.0001
Year×Status	1	2.5352	0.0643
Residual	390		
Total	399		

A total of 23 macroalgal taxa were enumerated within permanent plots for both 2006 and 2009 surveys. Understoreys were comprised of *Osmundaria colensoi Pterocladia lucida, Pterocladia capillacea, Carpomitra costata*, *Melanthalia abscissa Zonaria turneriana, Carpomitra* with crustose coralline algae (CCA), and coralline turf ubiquitous.

Analysis of biomass of the 23 macroalgal taxa enumerated within permanent plots indicated statistically significant differences between reserve and non-reserve areas (Status) and for the random factor Site(Status×Year), again reflecting high heterogeneity among sites within reserve and non-reserve areas between 2006 and 2009 (Table 3.4). Despite the increased biomass of several macroalagal taxa within many reserve and non-reserve sites in 2009, the factors Year and Year×Status were not statistically significant suggesting that macroalgal biomass has remained reasonably constant between 2006 and 2009 across protected and unprotected areas.

Table 3.4 Results from PERMANOVA (degrees of freedom, F ratios and P values from permutation) for biomass of macroalgae (23 taxa) from permanent quadrat sampling at reserve and non-reserve sites between 2006 and 2009 surveys.

Source	df	*F*	*P* (perm)
Year	1	1.6904	0.1164
Status	1	4.6688	0.0011
Site(Year×Status)	36	13.5458	0.0001
Year×Status	1	1.3382	0.2306
Residual	390		
Total	399		

Invasive algal taxa

Over the course of the survey no invasive algal species, such as the laminarian alga *Undaria pinnatifida* or the green alga *Caulerpa taxifolia* were observed within or adjacent permanent quadrats.

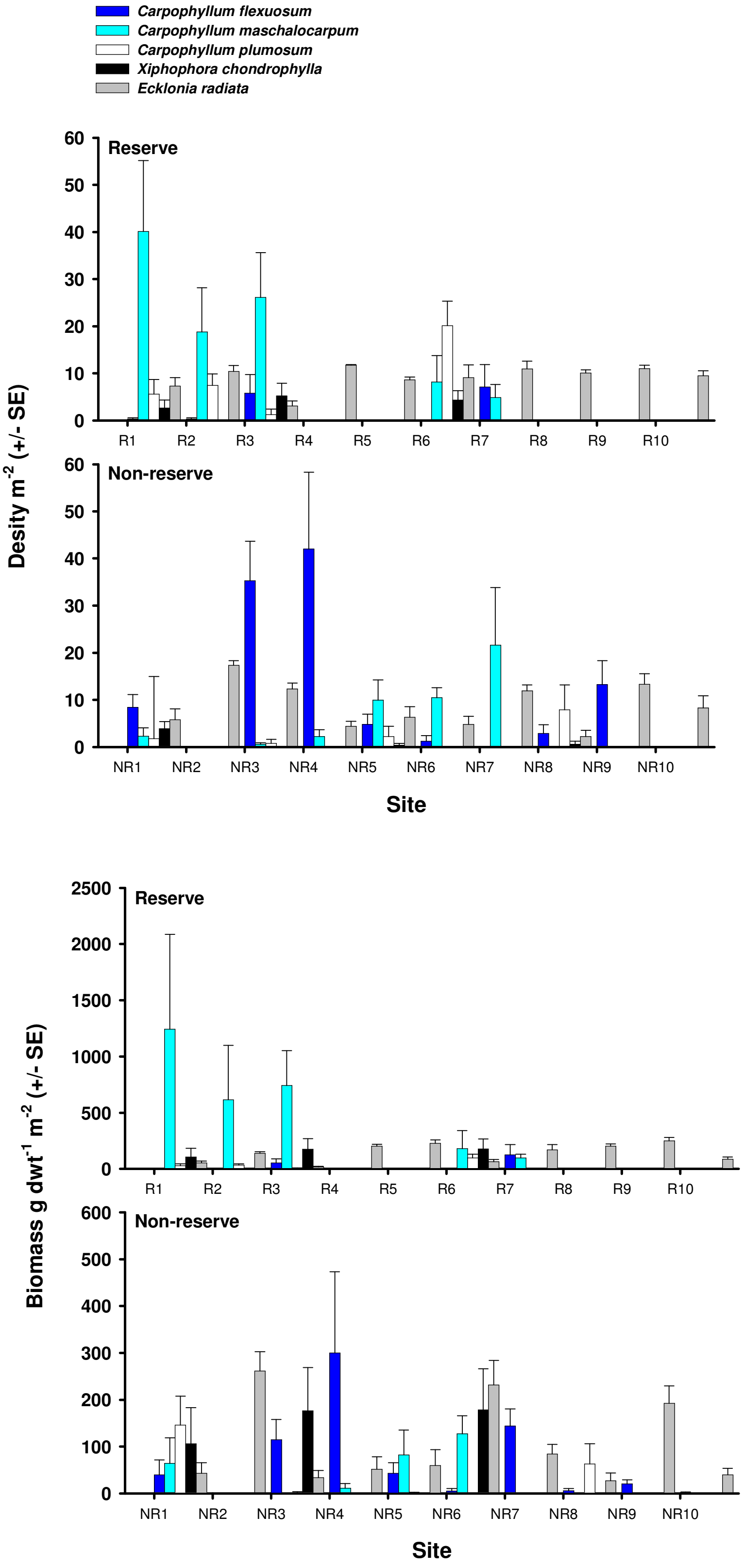

Figure 3.4. Mean density and biomass of dominant macroalgae within Te Whanganui-a-Hei Marine Reserve and non-reserve areas sampled. Refer to Fig. 2.1 for sampling sites.

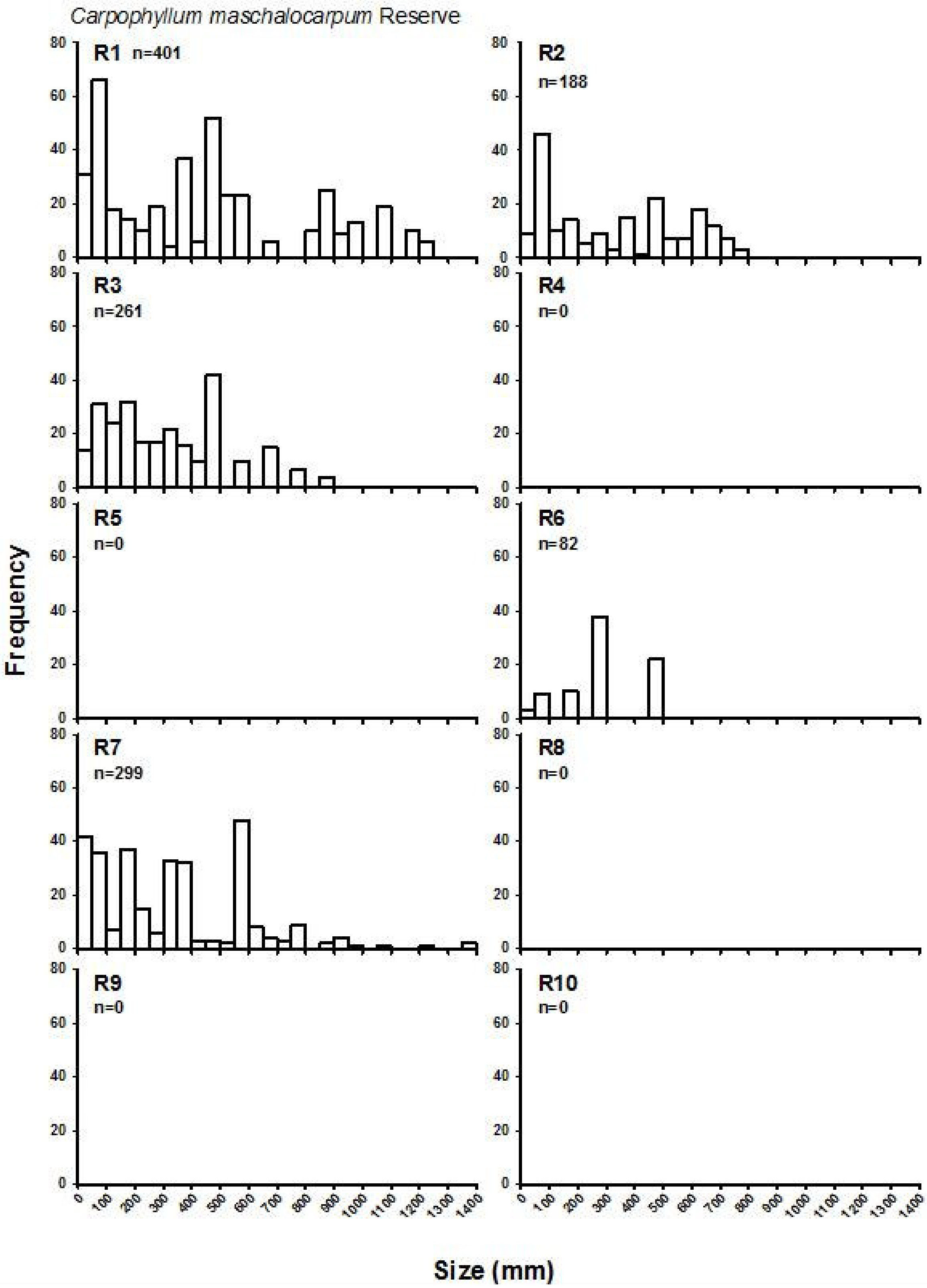

Figure 3.5. Size frequency distribution of *Carpophyllum maschalocarpum* within Te Whanganui-a-Hei Marine Reserve in 2009.

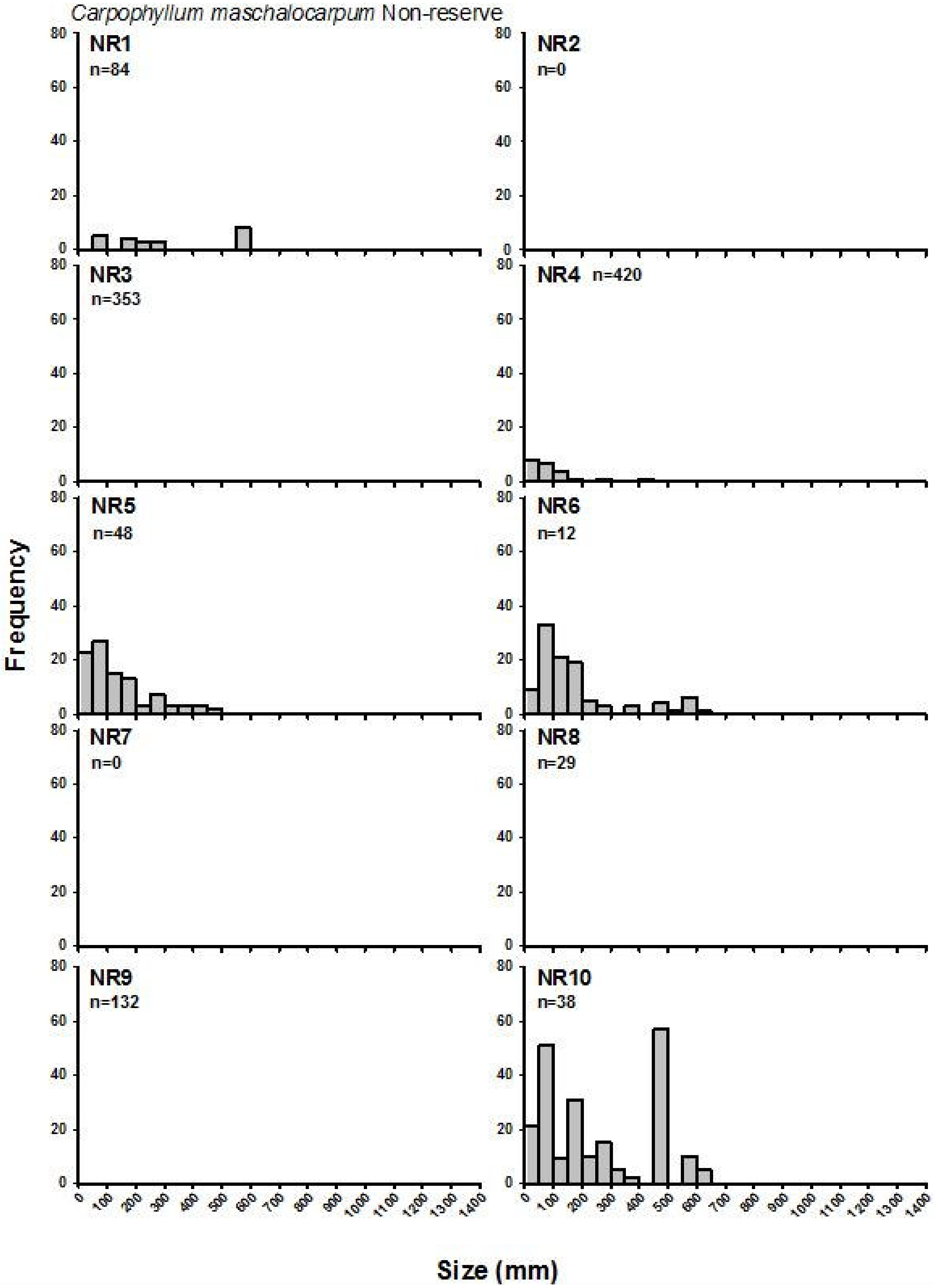

Figure 3.5 continued Size frequency distribution of *Carpophyllum maschalocarpum* within non-reserve sampling sites in 2009.

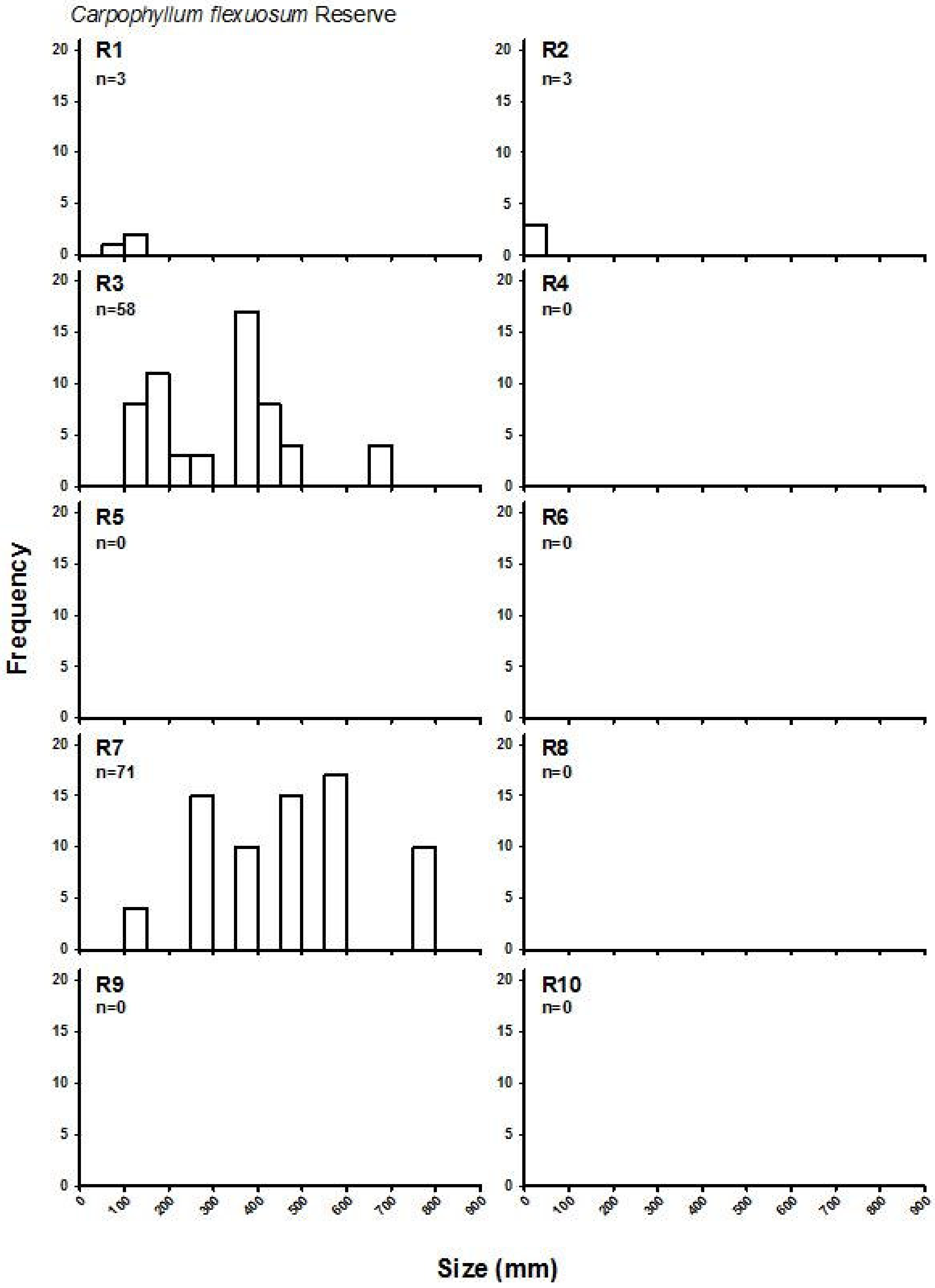

Figure 3.5 continued Size frequency distribution of *Carpophyllum flexuosum* within Te Whanganui-a-Hei Marine Reserve in 2009.

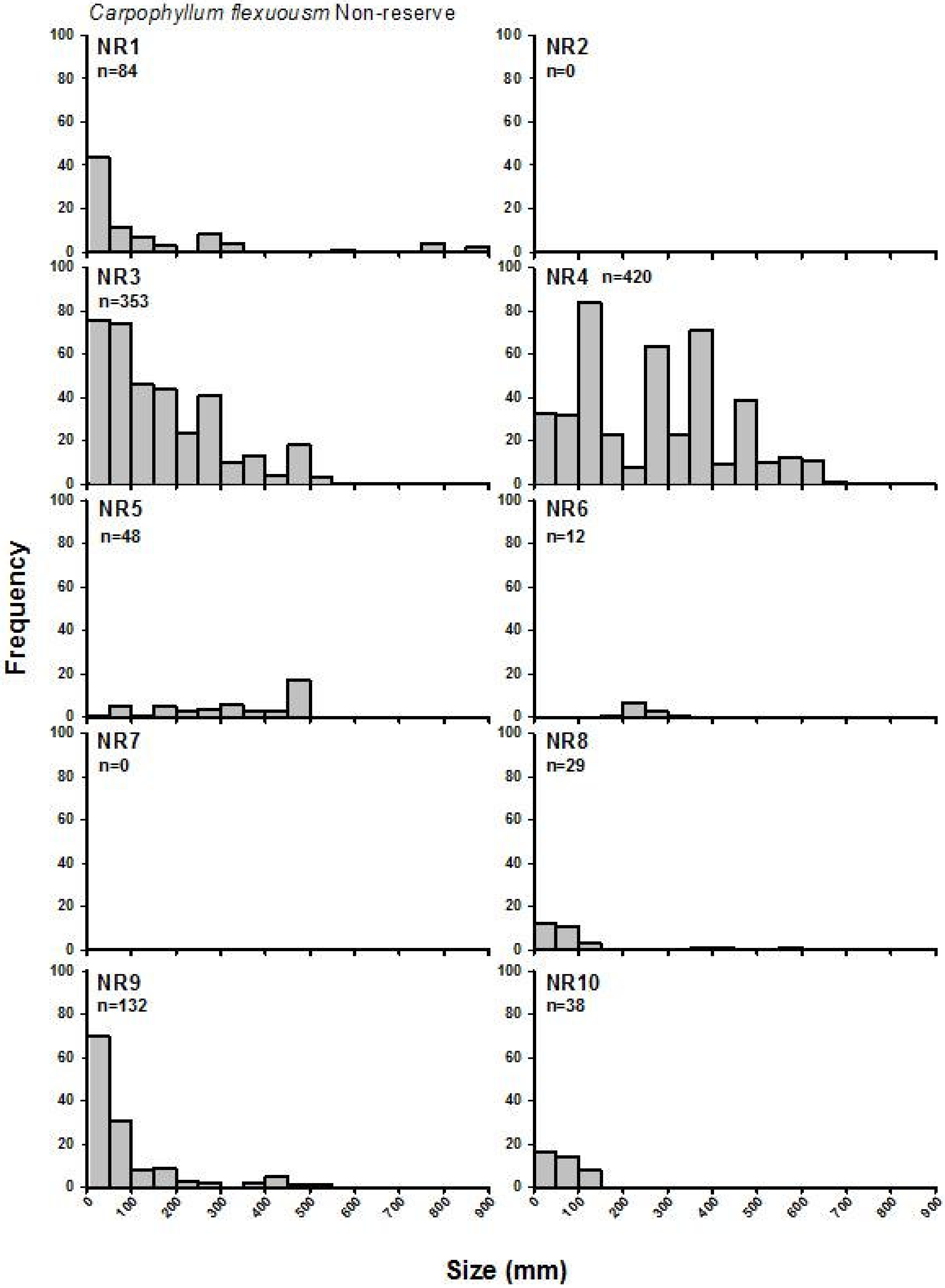

Figure 3.5 continued Size frequency distribution of *Carpophyllum flexuosum* within non-reserve sampling sites in 2009.

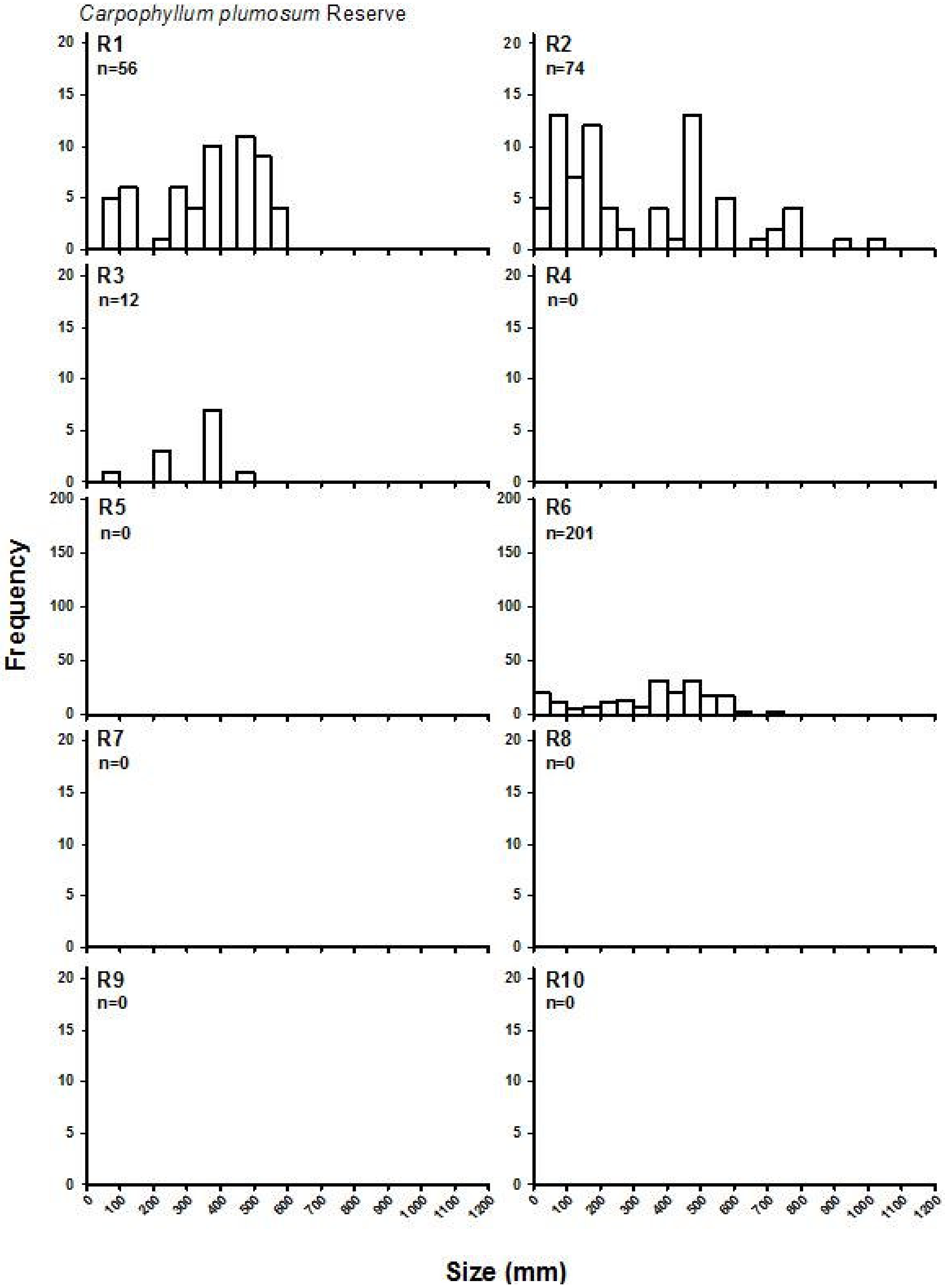

Figure 3.5 continued Size frequency distribution of *Carpophyllum plumosum* within Te Whanganui-a-Hei Marine Reserve in 2009.

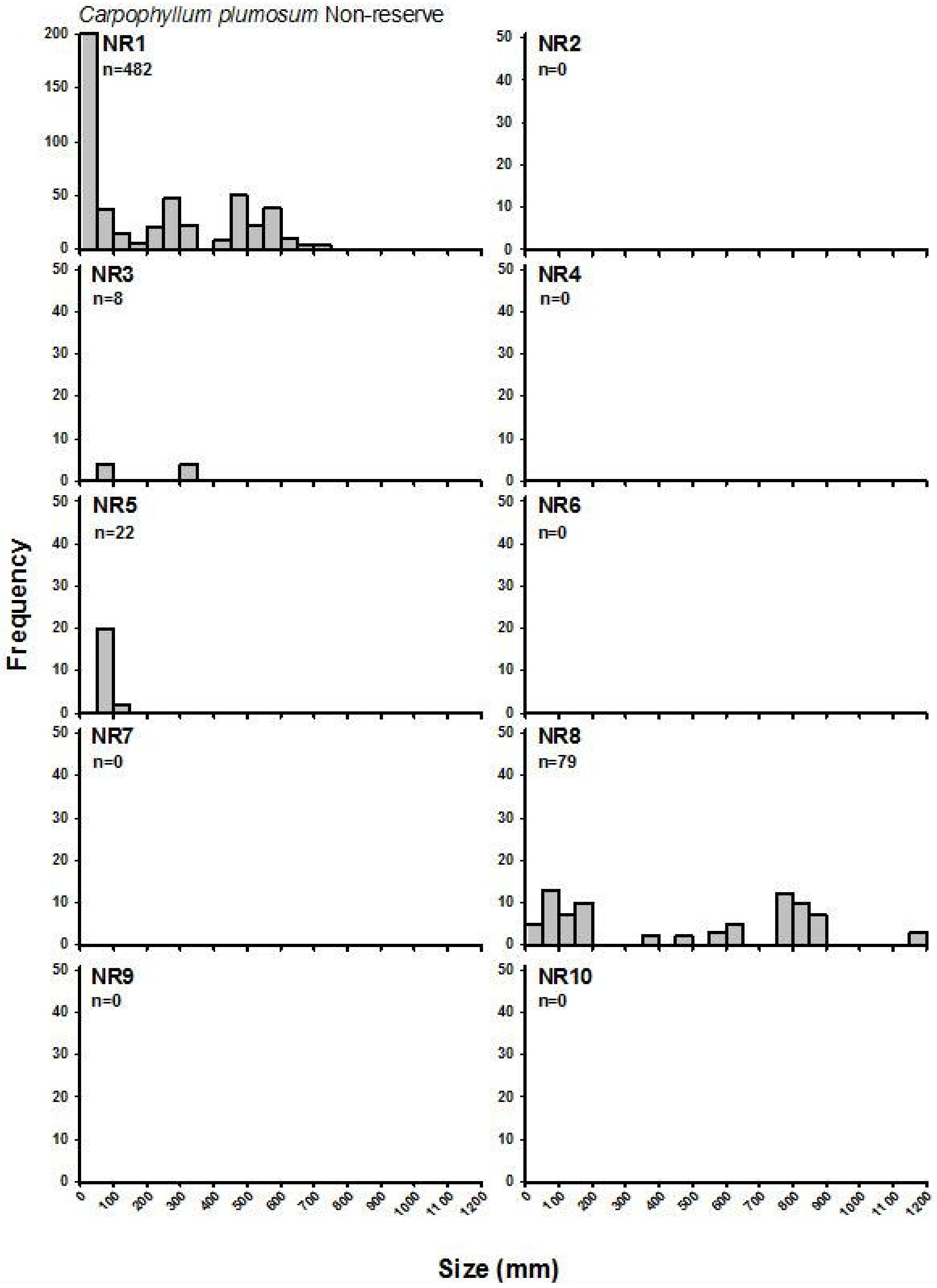

Figure 3.5 continued Size frequency distribution of *Carpophyllum plumosum* within non-reserve sampling sites in 2009.

MDS ordinations for 2009 macroalgal assemblages (23 taxa) based on biomass indicated three main groupings at the 60 % resemblance (similarity) level and are presented for the factors Status and Depth (Fig's 3.6). For the factor Status, all of the deep-water reserve sites (R4, R5, R6, R9 and R10) and two of the deep non-reserve sites (NR2 and NR10) were clustered to the right of the ordination. These sites were dominated exclusively by *Ecklonia radiata* and understorey algae such as *Carpomitra costata*, *Padina australis* and *Pterocladia capillacea*. Clustered to the left of the ordination were reserve shallow sites (R1, R3 and R6) and non-reserve shallow-water sites (NR1, NR3, and NR8), which with the exception of NR8 were all characterised by mixed algal assemblages dominated by fucalean algae, and diverse understorey taxa including *Zonaria turneriana, Osmundaria colensoi, Carpomitra costata* and *Pterocladia lucida*. *Ecklonia radiata* stands and mixed algal assemblages were also typical of sites clustered to the center of the ordination, e.g., R2, NR7, NR4, and NR6. Not surprisingly, the MDS ordination for the factor depth indicated that the cluster groupings for macroalgal biomass were strongly reflective of depth with sites located to the right of the ordination all deep-water sites, groups to the left all shallow water sites, with central group a mix of shallow and deep-water sites, i.e., sharing elements of the other two groups.

For the 2006 macroalgal biomass data, MDS ordinations indicated four clusters at the 60 % resemblance level (Fig. 3.7). Again, the factor Depth rather than Status appeared to be an important discerner of these groupings with deep-water sites clustered to the right of the ordination and shallow-water sites to the left. The tight clustering of many of the deep-water sites suggests a high degree of similarity in macroalgal assemblage structure, whereas the shallow-water sites from had a greater dissimilarity, reflecting the high variability of macroalgal assemblages in shallow-water. Shallow-water sites R1, R2, R3, R6 and NR1 clustered to the top of the ordination are all located in the northern region of the survey area and dominated by mixed-algal habitat, whereas sites NR3, NR4, NR5, NR7 and NR8, clustered to the bottom of the ordination, are all located to the south of the reserve were characterised by patchy mixed algal habitat and extensive urchin barrens habitat.

Further analysis using distance-based multivariate analysis (Distlm V2; Anderson 2002) to examine the importance of environmental variables in explaining the multivariate pattern of macroalgal biomass for 2006 and 2009 surveys strongly supported the MDS ordination for depth. Of the five factors tested – Depth, Wave Exposure, Sediment % Cover, Substratum Type, and Distance from the Whitianga Catchment – Depth was the only statistically significant factor (*Pseudo-F*=5.64, *P*=0.001) explaining 40 % of the variation in the 2009 dataset. Similarly, for 2006, Depth was the only statistically significant factor (*Pseudo-F*=8.09, *P*=0.001), explaining 31 % of the variation.

Note: Refer to CD-Rom for macroalgal biodiversity measures.

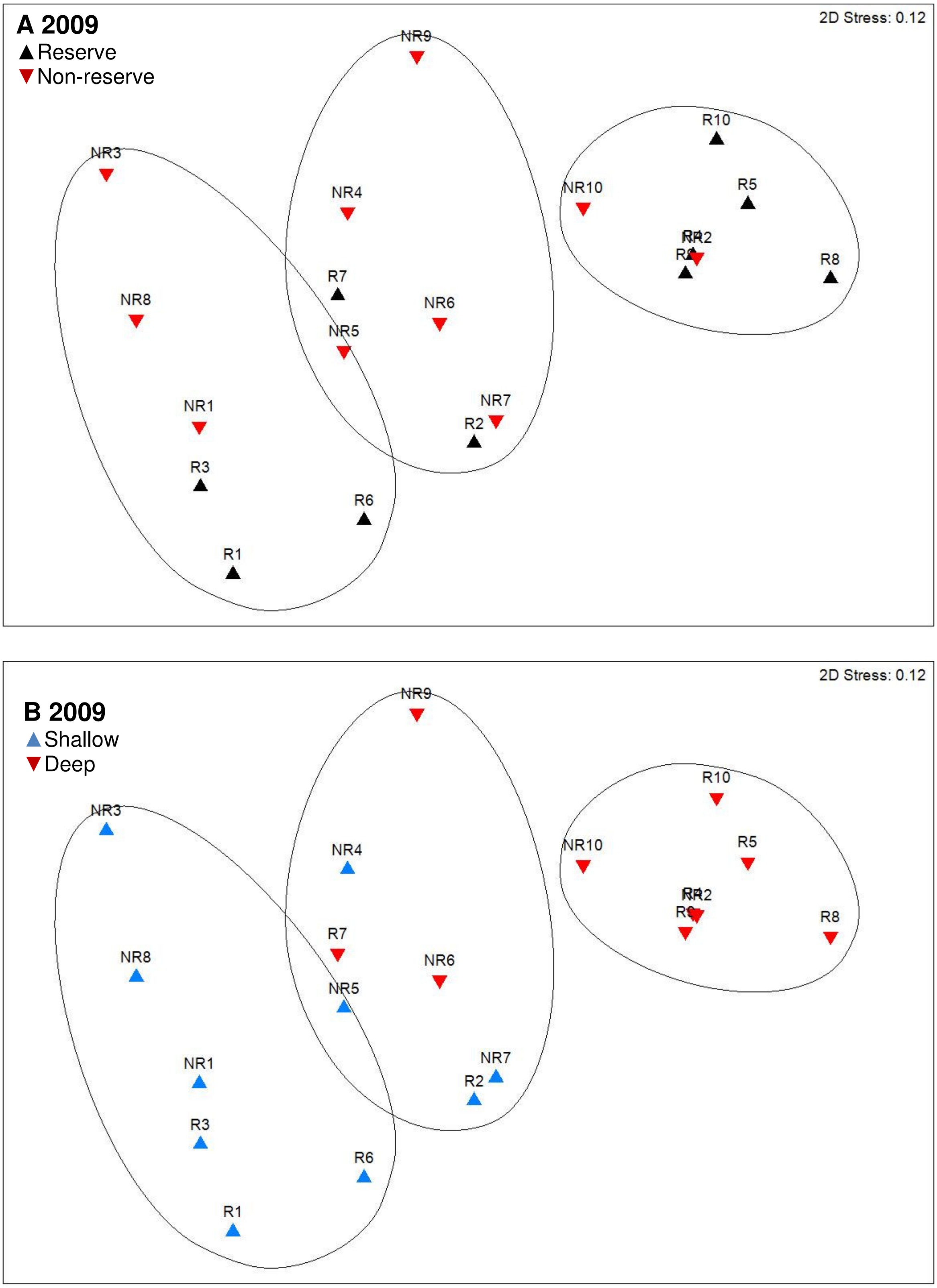

Figure 3.6. MDS ordinations for 2009 macroalgal biomass (23 taxa) across sites for A the factor status (reserve and non-reserve) and B the factor depth (shallow and deep). Clusters are depicted at the 60 % similarity level.

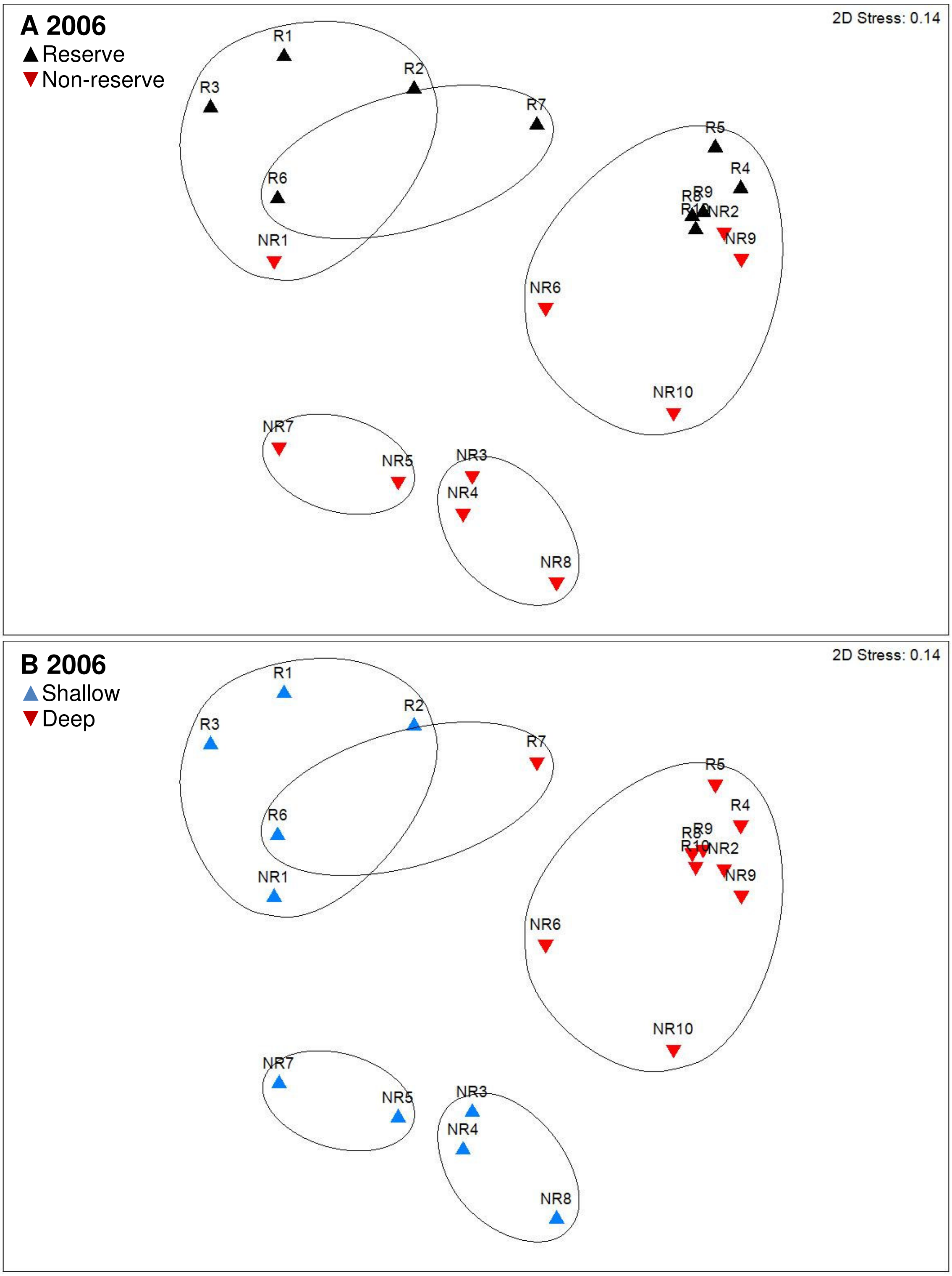

Figure 3.7. MDS ordinations for 2006 macroalgal biomass (23 taxa) across sites for A the factor status (reserve and non-reserve) and B the factor depth (shallow and deep). Clusters are depicted at the 60 % similarity level.

3.2 Substratum Cover

In 2009 crustose coralline algae (CCA) was again a dominant substratum cover across all reserve sites sampled, commonly > 50 % m^{-2} and generally increased in cover at many reserve sites (Fig. 3.8). Mirroring patterns within the reserve, CCA was the also dominant substratum cover at non-reserve sites in 2009 and 2006, often > 60 % m^{-2} (Fig. 3.8). Analysis (GLIMMIX) of CCA biomass suggested a statistically significant difference for the factor Status, with biomass higher across non-reserve sites than reserve sites. The non-significant factor Year suggests that CCA biomass has remained generally stable across years within the reserve and non-reserve areas surveyed, whereas the random effect Site(Status×Year) was highly significant, reflective of the high variability across sites surveyed (Table 3.2).

Coralline turf/articulated coralline algae was conspicuous at many of the reserve sites in 2009, but generally was < 30 % m^{-2} and similar to 2006 levels across sites (Fig. 3.9). Similarly, coralline turf /articulated coralline was a feature of the substratum at all non-reserve sites in 2009 and remained particularly dominant at NR7 and NR8 and increased in percent cover at sites NR3 and NR4 between 2006 and 2009 (Fig. 3.9).

In 2009, fine sediment was particularly noticeable on the substratum at site R5 comprising > 40 % of the substratum cover and was also evident at shallow-water sites R1, R2, and deep-water sites R4, R8, R9, R10 (Fig. 3.10). Fine sediment levels in 2009 were also generally higher than for 2006 at many of the reserve sites surveyed (Fig. 3.12). In deeper water, this may be reflective of the reduced *Ecklonia radiata* canopies, but may also be due to a combination of higher rain fall associated with La Niña climatic periods and the influence of the Whitianga catchment in this area of the reserve. For non-reserve sites surveyed fine sediment levels were typically < 5 % m^{-2} and much lower than that recorded within the reserve, but had generally increased compared to 2006 levels, particularly at sites NR7, NR8, and NR10 (Fig. 3.7).

Low lying deep rocky reef habitat most commonly associated with sites R4, R9, and R10 in the central region of the reserve also had a high to moderate cover of sand (Fig. 3.11). High sand cover in this region of the reserve is also likely related to oceanographic characteristics associated with La Niña conditions. Analysis of oceanographic data indicated higher wave energy was present from the east between May 2008 to May 2009, in comparison to May 2007 to May 2008, which also had greater wave activity than the period May 2006 to May 2007 (Fig. 3.13).

To verify the increase wave activity shown in the past 3 years of local wave data (Fig. 3.14), 12-years of offshore wave data were also considered. Figure 3.14 presents the long term wave climate offshore of Hahei, illustrating the percentage of wave height and direction of occurrence, whereas Figure 3.14 presents the wave roses for May 2008 to May 2009 and May 2007 to May 2008. It is clear that wave events from the north-east to south-east occurred with greater frequency and that calm periods were significantly less that those expected ‘on average’. Thus, the increased sand cover, especially of the heavier sand fractions within the deeper regions of the reserve, is likely to be the result of the past 2 years of La Niña conditions.

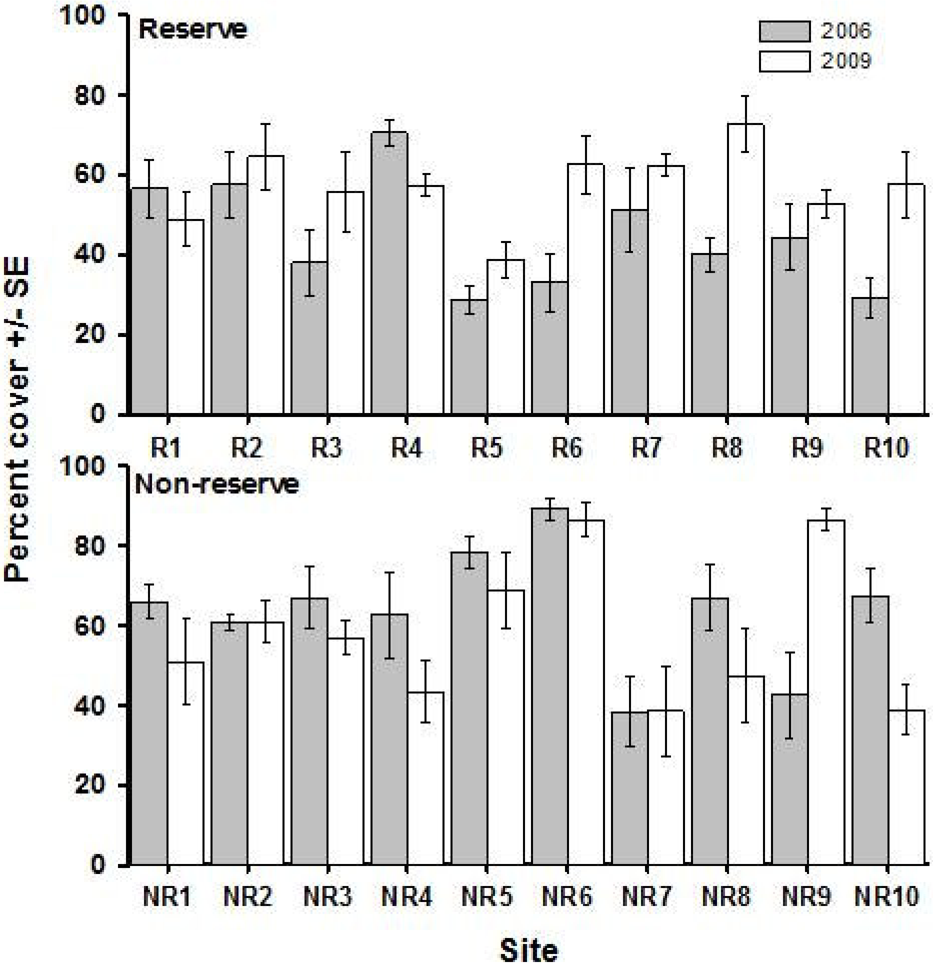

Figure 3.8. Percent cover of crustose coralline algae (CCA) within Te Whanganui-a-Hei Marine Reserve and non-reserve sites in 2006 and 2009.

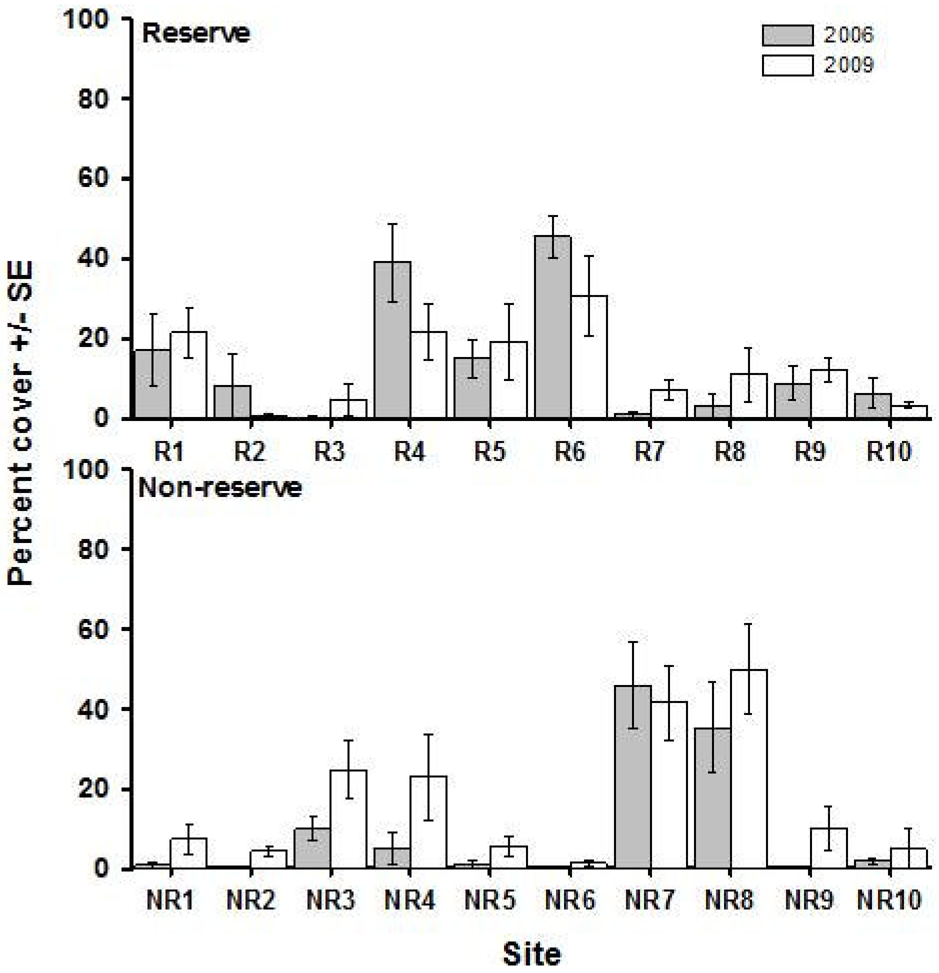

Figure 3.9. Percent cover of coralline turf/articulated coralline within Te Whanganui-a-Hei Marine Reserve and non-reserve sites in 2006 and 2009.

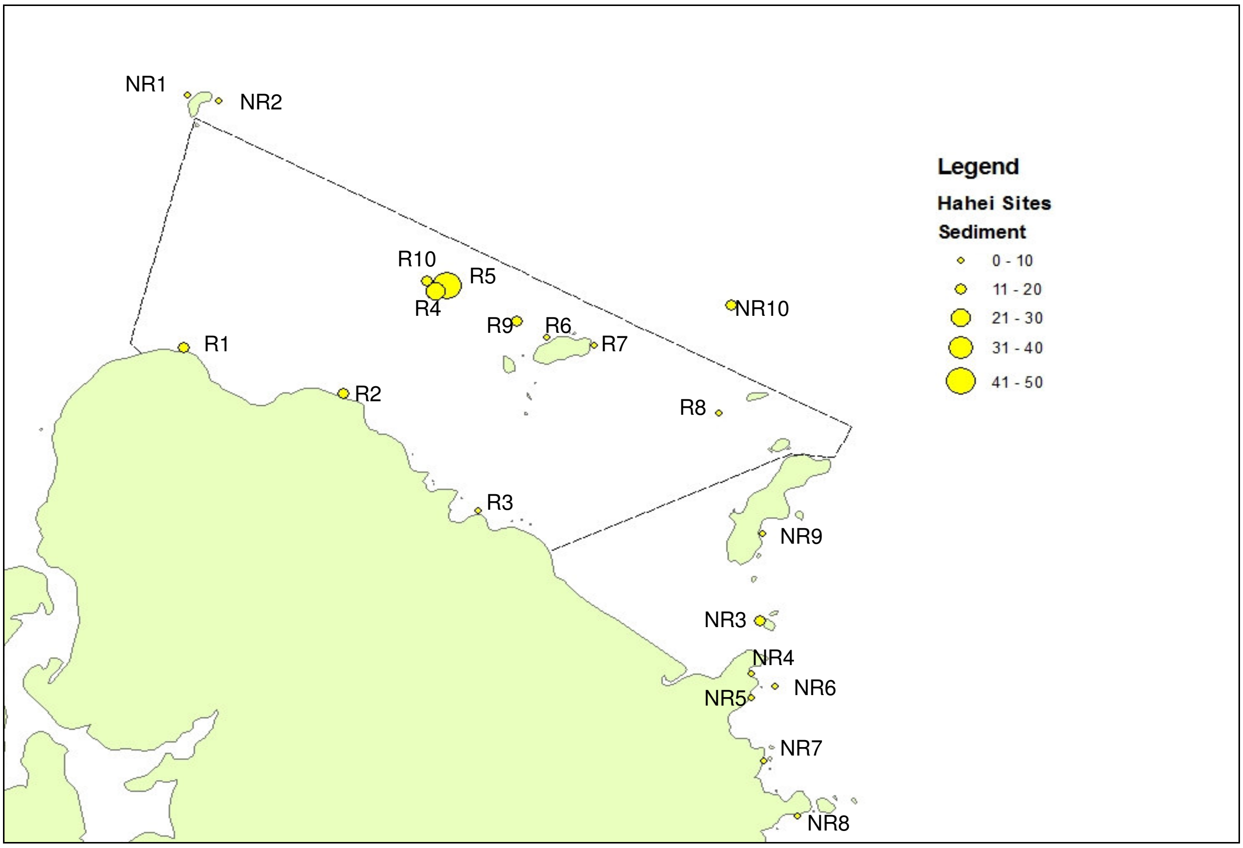

Figure 3.10. Average percent cover of fine sediment for reserve (R) and non-reserve (NR) sites surveyed in 2009.

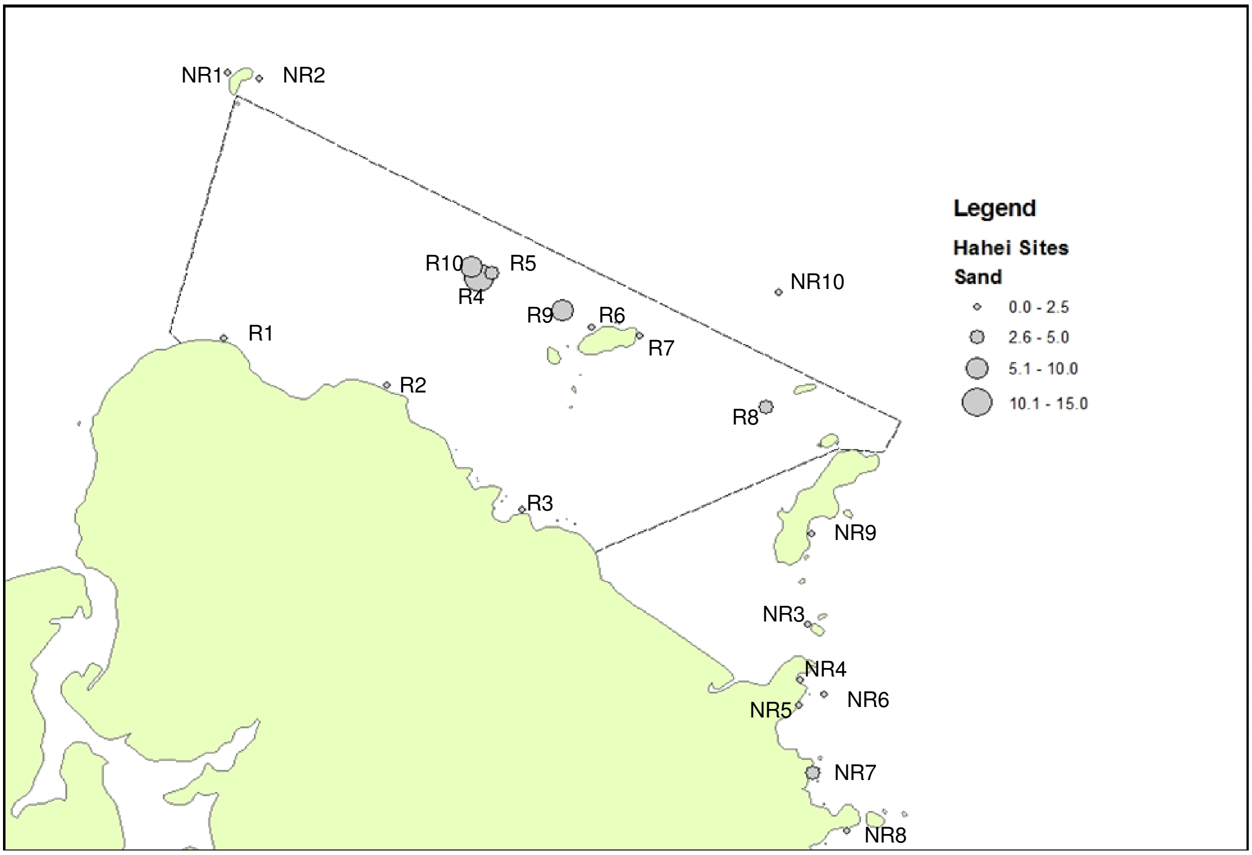

Figure 3.11. Average percent cover of sand for reserve (R) and non-reserve (NR) sites surveyed in 2009.

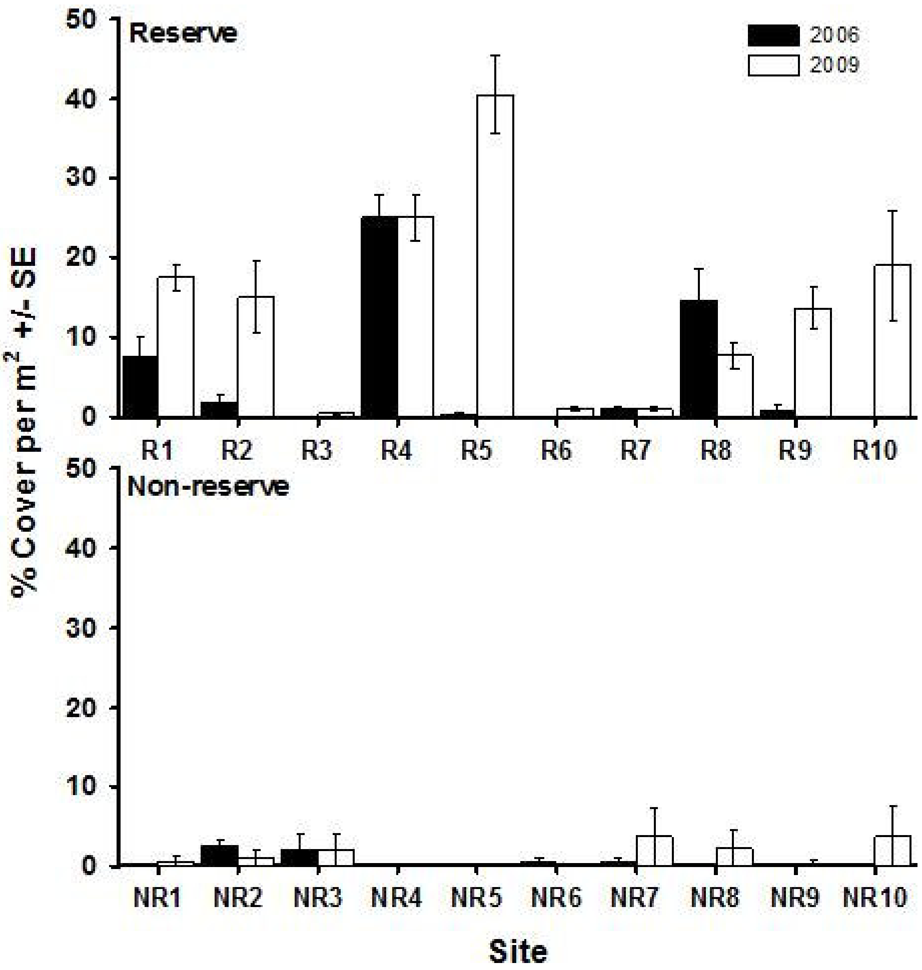

Figure 3.12. Percent cover of sediment within Te Whanganui-a-Hei Marine Reserve and non-reserve sites in 2006 and 2009.

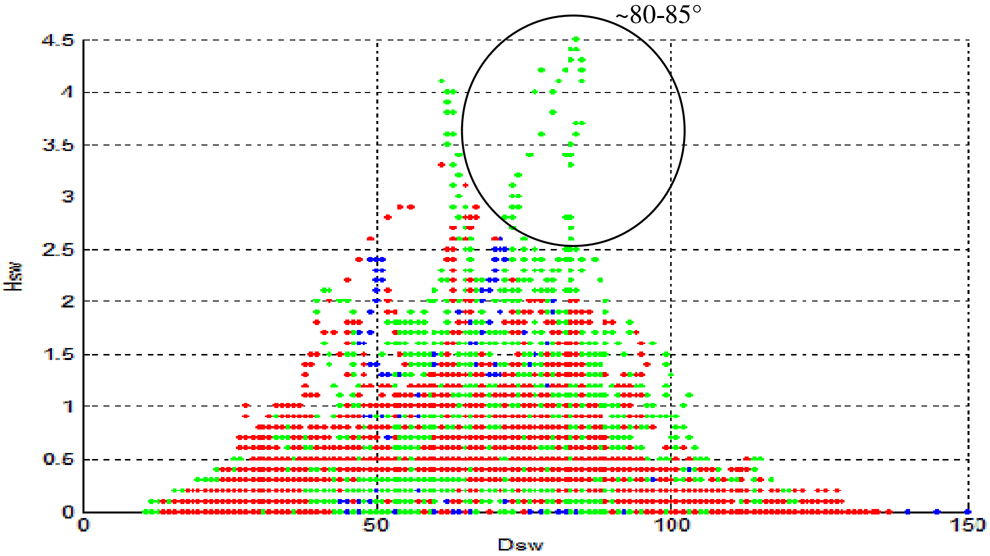

Figure 3.13. Marine weather (www.marineweather.co.nz) archive of swell height by direction for Hotwater Beach (south of the reserve). Blue = 2006-07, Green = 2007-08, Red = 2008-09.

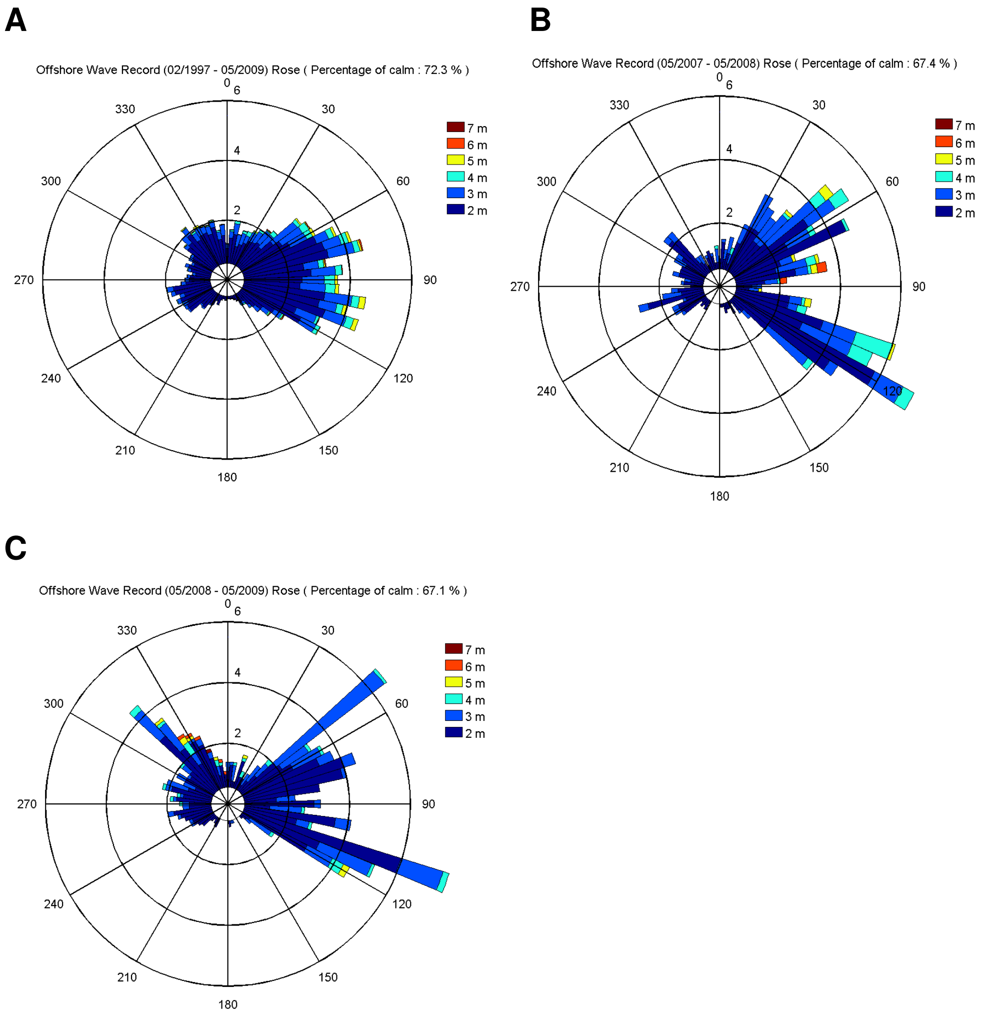

Figure 3.14. Wave roses of wave height and direction for A: 1997 - present, B: May 2007-May 2008 and C: May 2008-May 2009.

3.3 Sessile Invertebrates

A total of 54 sessile invertebrates were enumerated across reserve and non-reserve sites surveyed in 2006 and 2009. Taxa richness for 2009, and mean taxa richness for 2006 and 2009 are presented in Fig's 3.15 & 3.16. Generally, deep-water reserve sites had higher diversity than shallow-water sites, a pattern that was consistent between surveys. Mean sessile invertebrate diversity was highest (> 8 per m^2) at sites R4, R8 and R9. For the remaining sites (a mix of shallow and deep) mean taxa richness was commonly < 5 per m^2. Deep-water non-reserve sites NR2 and NR10, adjacent the offshore reserve boundary also had higher taxa richness and mean taxa richness (> 8 per m^2) than the remaining non-reserve shallow-water sites surveyed (Fig. 3.15, 3.16).

Multivariate analysis using PERMANOVA for percent cover data indicated that sessile invertebrate assemblages were statistically different between surveys (Year), reserve and non-reserve areas (Status), being higher within the reserve, and for the random factor Site(Year×Status) reflecting the high variability in sessile invertebrate assemblages across sites between surveys (Table 3.5). Analogous to macroalgal assemblage patterns, MDS ordinations for the 2009 and 2006 survey data for Status and Depth (Fig's 3.17, 3.18) suggested that sessile invertebrate assemblage groupings are reflective of depth-related patterns, rather than protection status *per se*. Two main groupings were discernable at the 40% resemblance level for both the 2009 and 2006 surveys with deep-water sites clustered to the left of the ordination and shallow-water sites to the right of the ordination. Two deep water sites NR6 and R7 were more-similar to the majority of shallow-water sites for both surveys, and interestingly these two sites were the shallowest of the deep-water sites surveyed (approximately 10 m depth), whereas the other deep-water sites surveyed were greater than 12 m depth.

Table 3.5. Results from PERMANOVA (degrees of freedom, *F* ratios and *P* values from permutation) for sessile invertebrates (51 taxa) from permanent quadrat sampling at reserve and non-reserve sites between 2006 and 2009 surveys.

Source	df	F	P (perm)
Year	1	3.7850	0.0009
Status	1	2.8023	0.0083
Site(Year×Status)	36	3.8589	0.0001
Year×Status	1	1.3142	0.2111
Residual	390		
Total	399		

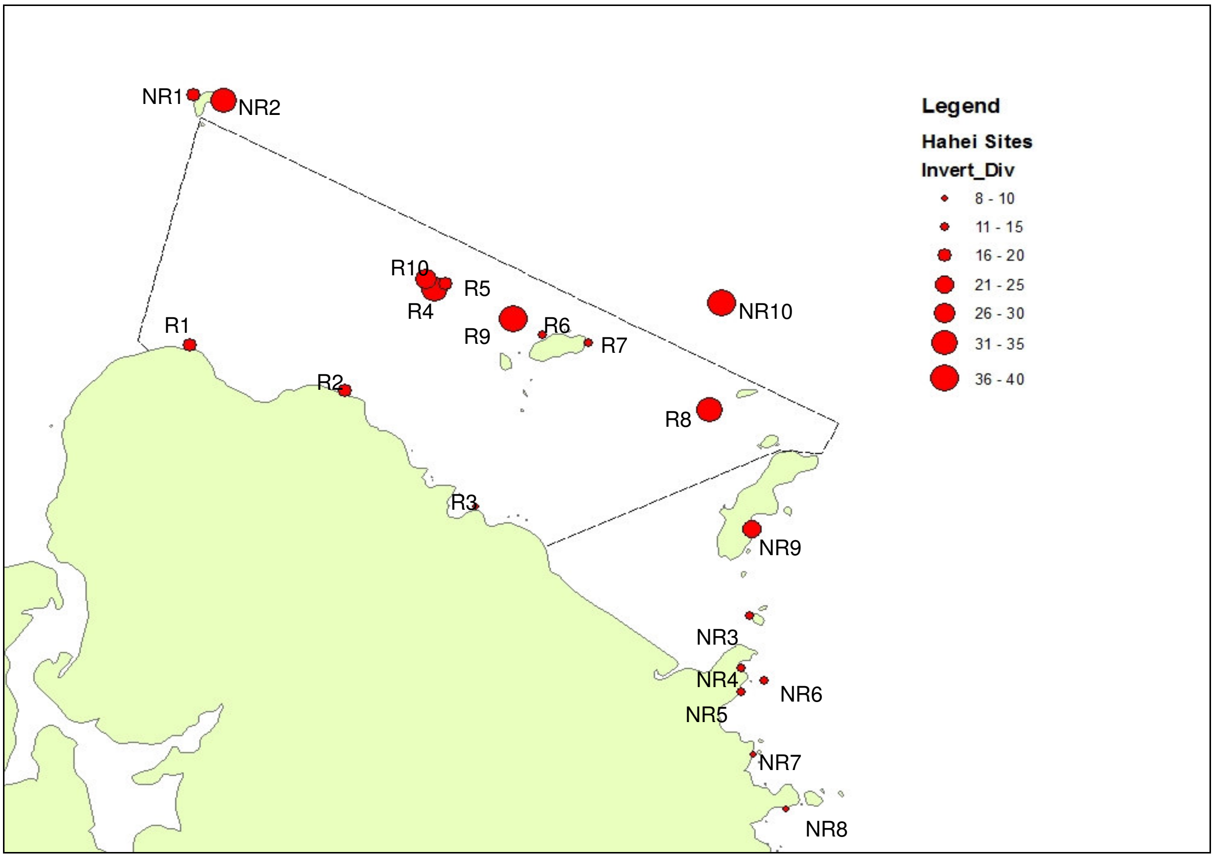

Figure 3.15. Sessile invertebrate taxa biodiversity based on taxa richness (S) for reserve (R) and non-reserve (NR) sites surveyed in 2009. Refer to CD-Rom for additional biodiversity measures.

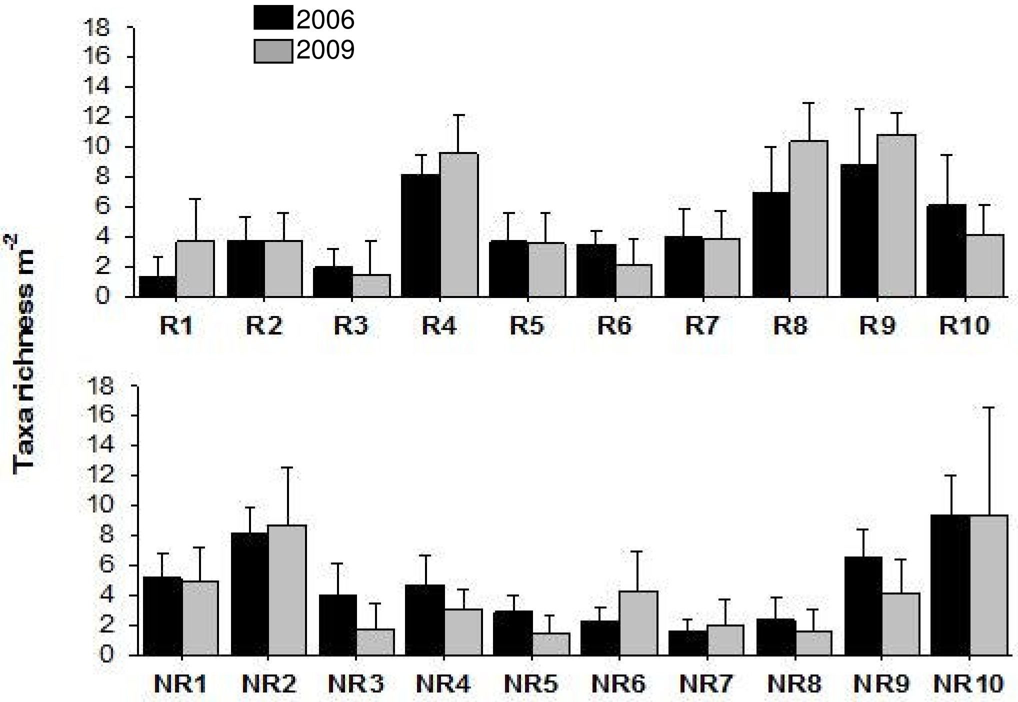

Figure 3.16. Mean sessile invertebrate richness for reserve and non-reserve sites surveyed in 2006 and 2009.

Differences between invertebrate assemblages for the factor depth were due to key invertebrate taxa that were only found at deep-water sites. For example, the sponges *Ancorina alata*, *Raspailia topsenti, Stelletta maori Psammocinia hawere.*, *Geodia* sp., *Callyspongia ramosa, Ircinea* sp, *Polymastia fusca, Polymastia aurantium*; ascidians *Didemnum maculatum* and *D. vexillum*; the soft coral *Alcyonium* sp; thecate hydroids *Sertularella* sp, and *Aglaophenia laxa*; and, bryozoans *Steginoporella* and *Hornera* sp.

Many taxa were also common across the majority of sites sampled irrespective of depth such as the colonial hard coral *Culicia rubeola* and solitary coral *Monomyces rubrum*, the golfball sponges *Tethya aurantium*, *Tethya ingalli* and *Tethya* sp, solitary sea squirts *Cnemidocarpa bicornuta* and *Asterocarpa coerulea*, the ascidian *Pseudodistoma novaezelandiae* and the anemone *Actinothoe albocincta*. With the exception of *Aplidium*, there were no taxa clearly unique to shallow-water sites, which were comprised of common taxa also found in deep water. These assemblage patterns were also typical of the sites surveyed in 2006. Selected bubble plots for average percent cover in 2006 and 2009 among sites for *Raspailia*, (abundant only at deepwater sites), *Tethya ingalli* (widespread across all sites, irrespective of depth) and *Aplidium* sp (dominant in shallow-water for the 2009 survey) are presented in Fig. 3.19.

Analysis of sessile invertebrate percent cover using distance-based multivariate analysis (Distlm V2) to examine the influence of environmental variables in explaining the multivariate patterns in 2006 and 2009 complimented MDS ordinations. For the 2009 survey, of the 5 variables tested (Depth, Wave Exposure, Sediment % Cover, Substratum Type, and Distance from the Whitianga Catchment) only the factor Depth was statistically significant (*Pseudo-F*=5.64, *P*=0.001) and explained 25 % of the variation in the dataset. Similarly, for the 2006 survey Depth was the only statistically significant factor (*Pseudo-F*=4.35, *P*=0.001 and explained approximately 22 % of the variability. The other variables tested were not statistically significant.

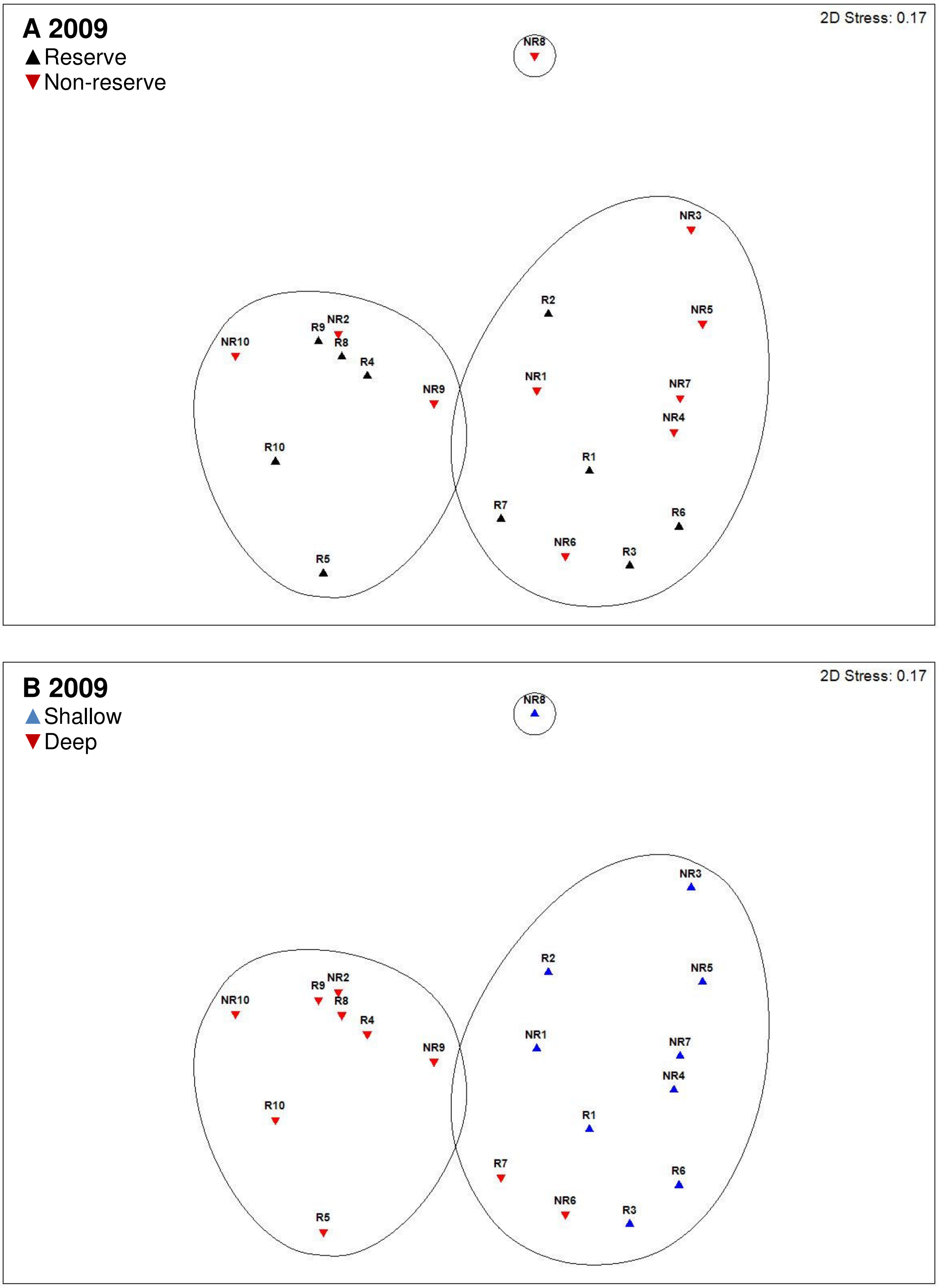

Figure 3.17. MDS ordinations for 2009 sessile invertebrate taxa assemblages across sites for A the factor status (reserve and non-reserve) and for B the factors shallow and deep. Clusters are depicted at the 40 % similarity level.

Figure 3.18. MDS ordinations for 2006 sessile invertebrate taxa assemblages across sites for A the factor status (reserve and non-reserve) and for B the factors shallow and deep. Clusters are depicted at the 40 % similarity level.

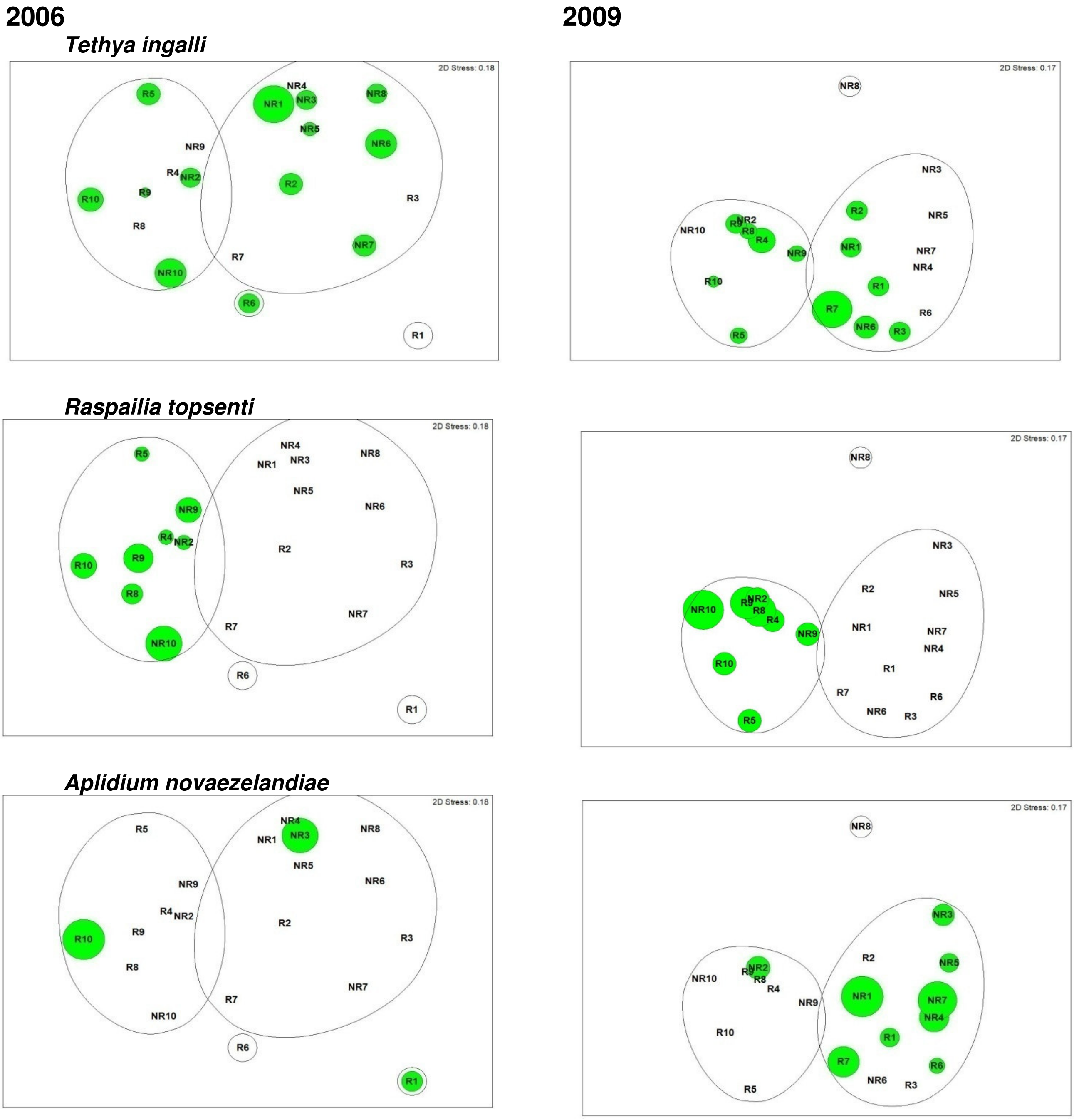

Figure 3.19. Selected bubble plots for *Tethya ingalli*, *Raspailia topsenti* and *Aplidium novaezelandiae* for 2006 and 2009 surveys. Individual plots indicate presence at a particular site (green bubble) and % cover (size of bubble).

3.4 Mobile Invertebrates

Evechinus chloroticus

Evechinus chloroticus was present within all shallow-reef sites surveyed within the reserve in 2006 and 2009 (Fig. 3.20). Highest densities in 2009 (> 5m^{-2}) occurred at R1 (Cooks Bluff) and R3 (Gemstone north, shallow), with mean abundance at R2 (Boulder Bank) and R6 (shallow Motueka Island) < 4 m^{-2}. *Evechinus chloroticus* also occurred at all shallow-water and several deep-water (NR9, NR10) non-reserve sites in 2006 and 2009. In 2006 highest densities occurred at NR5 and NR8 > 10 m^{-2} associated with barrens habitat, although 2009 data indicate reduced abundance at these sites. At site NR1 and the two deep-water sites NR9 and NR10 urchin abundance in 2009 had increased relative to 2006 levels. Overall, urchin abundance remains higher across the non-reserve survey area compared to the reserve and this difference was statistically significant (GLIMMIX, Table 3.6). The factor Year and interaction Year×Status were not statistically significant, although the random affect Site(Status×Year) was highly significant, reflecting the high variability in abundance levels among reserve and non-reserve sites between surveys.

Abundance patterns based on behavioural characteristics indicated that cryptic urchins, present at all shallow-water reserve sites, were consistently more-abundant than exposed urchins (Fig. 3.21). Cryptic urchin abundance in 2009 was higher at R1, R2 and R6 than in 2006, whereas abundance levels at R3 were broadly similar between surveys (Fig. 3.21).

Size frequency distribution comparisons (pooled across reserve) sites indicate that a greater frequency of larger urchins were cryptic in 2009 compared to 2006, and that the majority of smaller urchins censused < 80 mm TD were generally cryptic and this latter pattern was consistent between surveys (Fig 3.9). The mean size of both exposed and cryptic *E. chloroticus* within the reserve was also higher in 2009 than 2006 (Fig. 3.22).

The density of cryptic and exposed urchins across non-reserve sites was highly variable in 2009. Sites NR4 and NR5 had higher densities of cryptic urchins than exposed, which is a marked contrast to 2006 levels, where exposed urchins occurred at higher density at these sites. Sites NR5 and NR8 also had much lower densities of urchins (irrespective of behavioural status) in 2009, compared to 2006 levels where mean abundance was >10 m^{-2} (Fig. 3.21). *Evechinus chloroticus* also occurred at NR10 in 2009, which is surprising given that the permanent 100 m^2 quadrat at this site has an average depth of 13.1 m.

As for the reserve sample population, there were higher frequencies of larger (> 90 mm TD) cryptic urchins in 2009 than 2006 for non-reserve areas, although the general pattern of higher frequencies of exposed urchins > 80mm TD was maintained (Fig. 3.22). Size data also indicate an increase in the mean size of both exposed and cryptic urchins between 2006 and 2009 Fig. 3.23).

Evechinus chloroticus size, based on test diameter in 2009 (pooled across sites and behavioural class) was higher in non-reserve areas than reserve areas, and 2009 size was higher within non-reserve and reserve areas in 2009 compared to 2006. Consequently size was statistically significant (Two-Way ANOVA) for the factor Status F=21.2, P<0.001 and Year (F=65.2, P<0.001) with a non-significant Status×Year interaction (F=5.64, P=0.278) indicating rates of growth were similar between reserve and non-reserve populations through time.

Evechinus chloroticus sample populations within reserve and non-reserve sites in 2009 were all healthy (Health category 1 – see methods). This was also true for the 2006 survey, with the exception of site NR4 where 65 % of urchins had folded guard spines and 5 % had visible guard spine loss, typified by bald areas on the test (Health category 2).

Table 3.6. Results from mixed model analysis (GLIMMIX) for urchin (*Evechinus chloroticus*) abundance within Hahei reserve and non-reserve sites between 2006 and 2009. Significance: *p < 0.05, **p < 0.01, ***p < 0.001. Note: the Status×Year effect was non-significant.

Factor	**Fixed effect**		**Covariance parameter estimate**
Evechinus chloroticus	**Status**	**Year**	**Site (Status×Year)**
	$F_{1,18} = 6.45$**	$F_{2,36} = 1.73$	Z value = 5.20, Pr $Z < 0.0001$

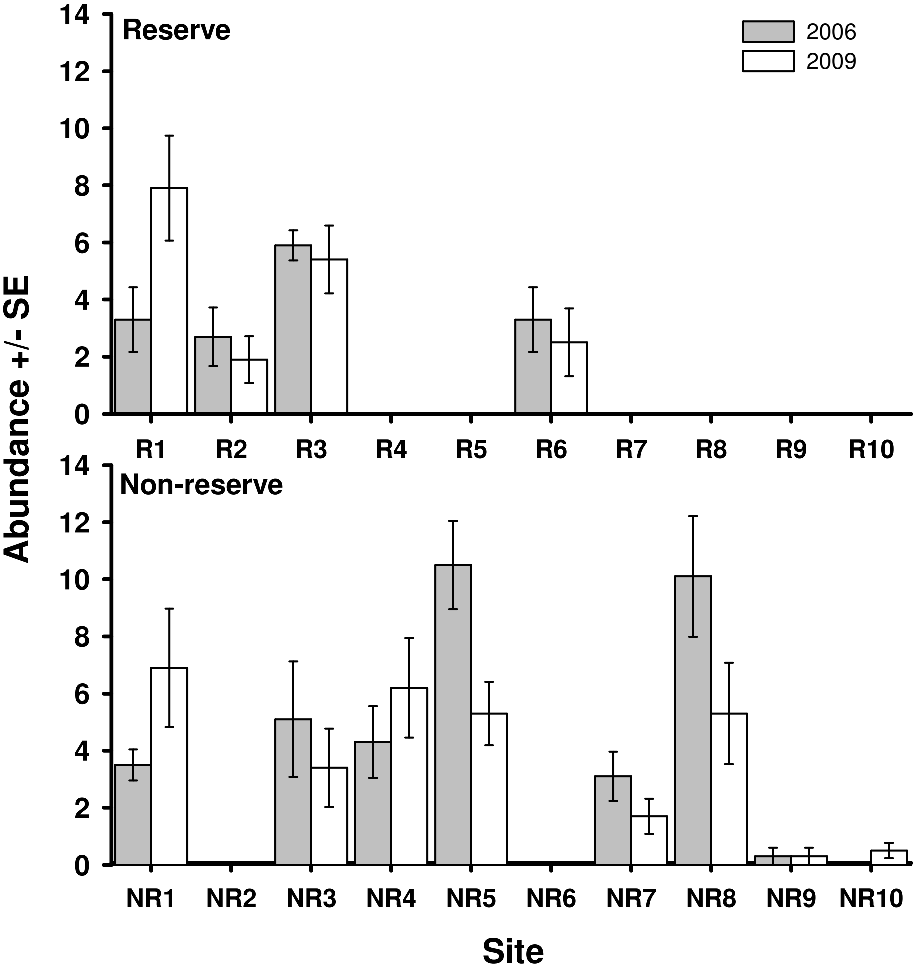

Figure 3.20. Density of *Evechinus chloroticus* within Te Whanganui-a-Hei Marine Reserve and non-reserve sampling sites in 2006 and 2009.

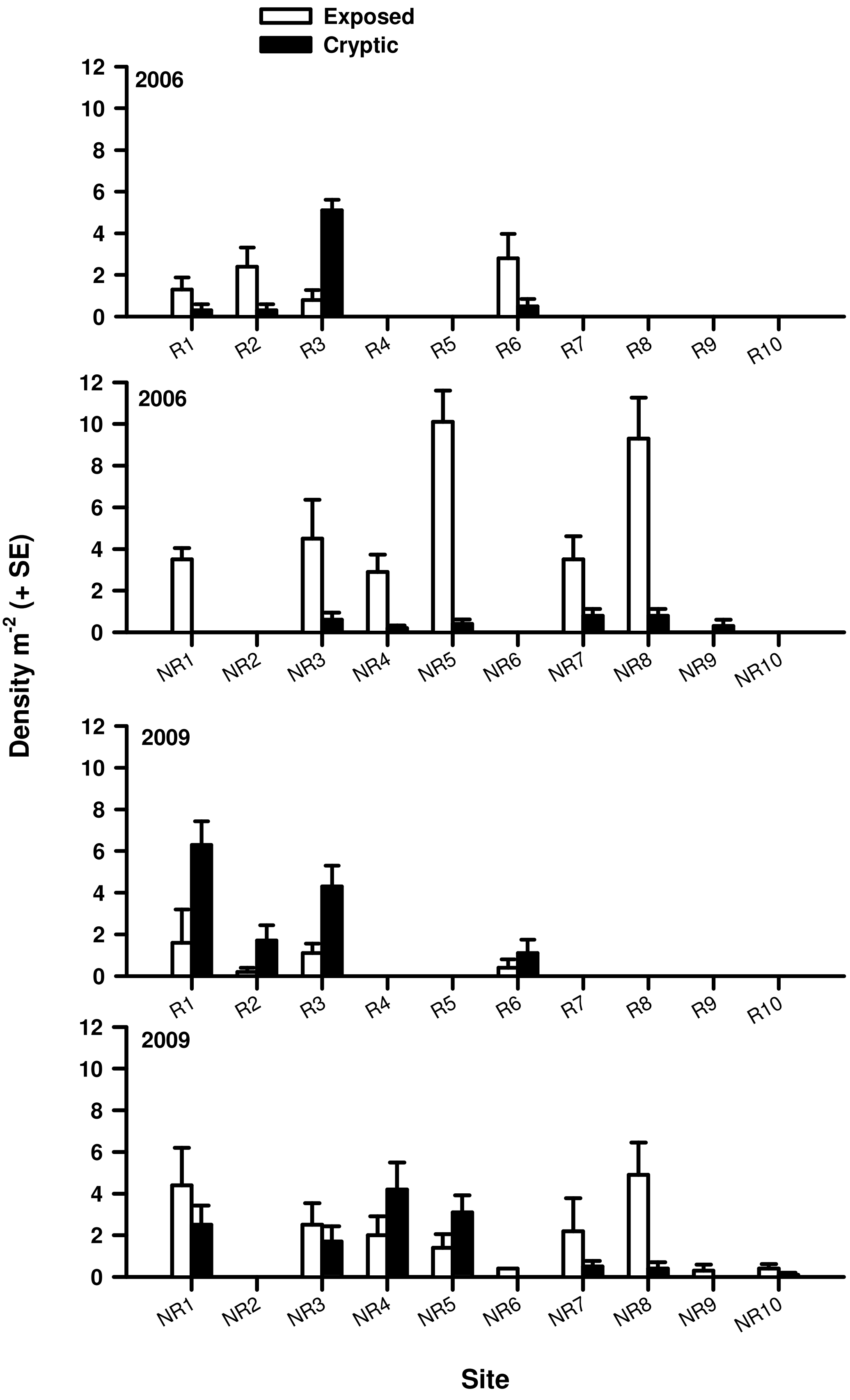

Figure 3.21. Density of exposed and cryptic *Evechinus chloroticus* within Te Whanganui-a-Hei Marine Reserve and non-reserve sampling sites in 2006 and 2009.

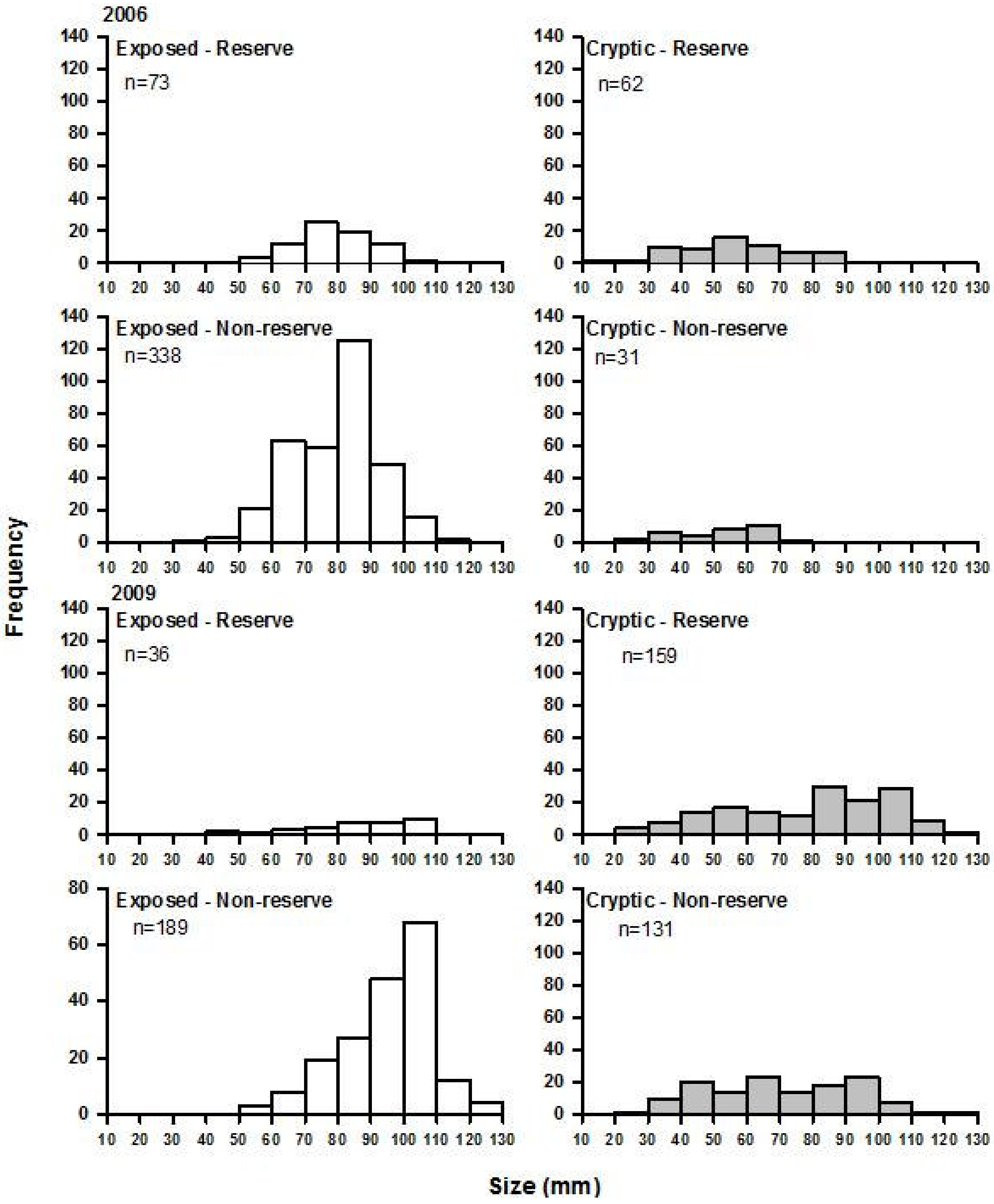

Figure 3.22. Size frequency distributions (based on test diameter) of exposed and cryptic *Evechinus chloroticus* within Te Whanganui-a-Hei Marine Reserve and non-reserve areas (pooled across sites) in 2006 and 2009.

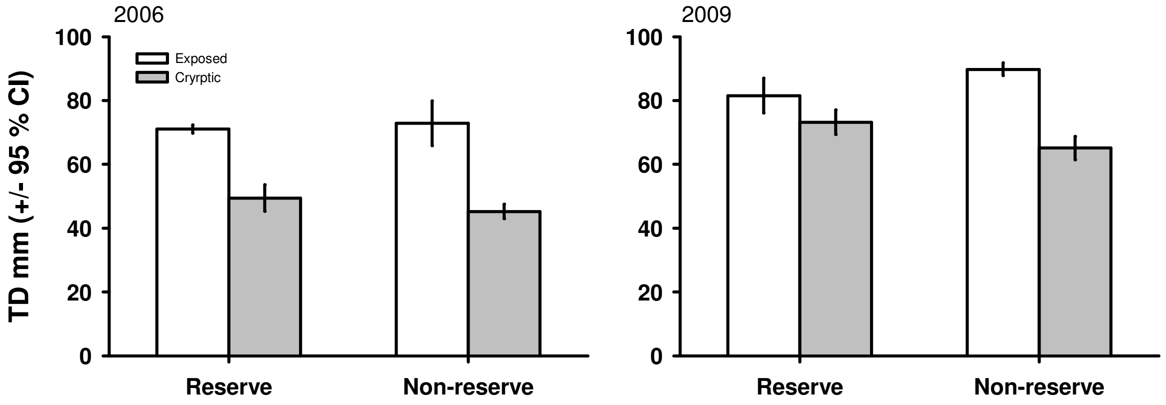

Figure 3.23. Test diameter of exposed and cryptic *Evechinus chloroticus* within Te Whanganui-a-Hei Marine Reserve and non-reserve areas (pooled across sites) in 2006 and 2009.

Molluscs

A total of 35 mollusc taxa were identified across reserve and non-reserve permanent plots in 2009 (refer to Appendix 2). Taxa richness ranged from 5 to 15 across sites with no clear pattern with depth or protection status (Fig. 24). As for 2006 (refer to Appendix 1), dominant taxa included the cooks turban *Cookia sulcata*, the green top-shell *Trochus viridis*, the opal top shell *Cantharidus purpureus,* and the granose turban *Modelia granosa*. Abundance of these taxa in 2009 was highly variable among sites and between reserve and non-reserve locations.

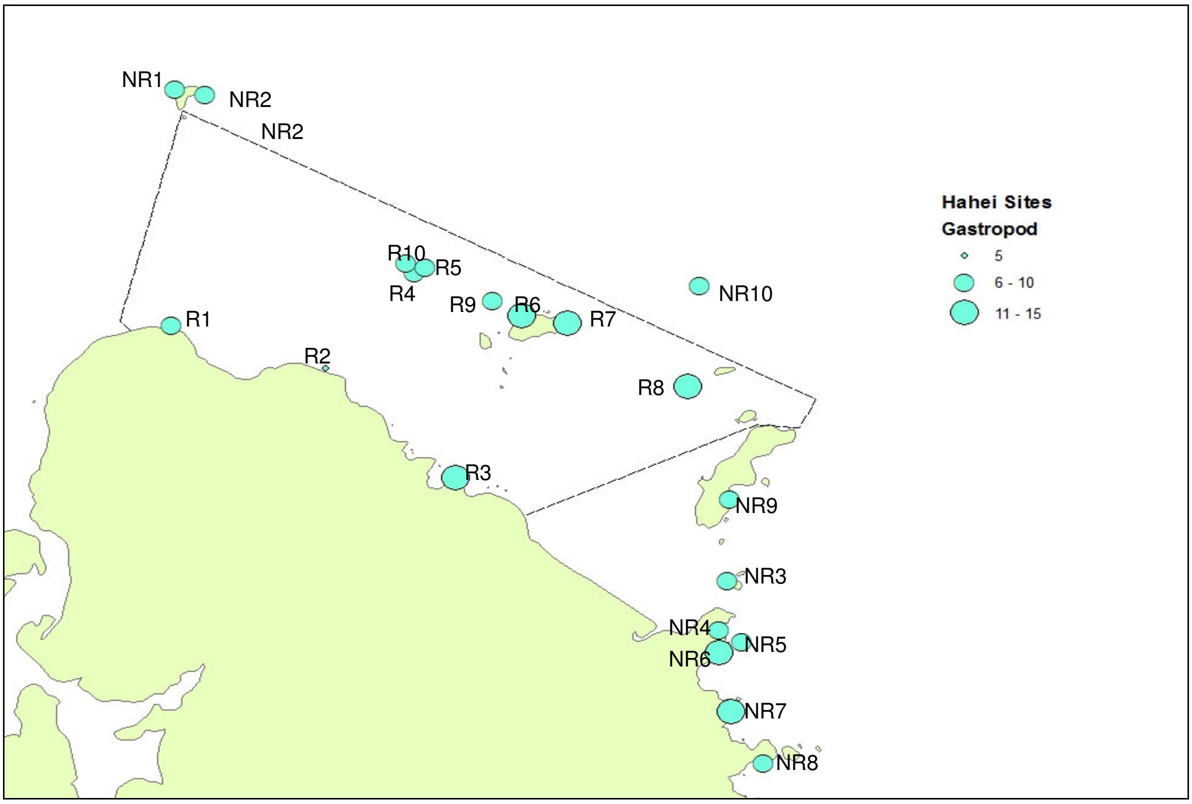

Figure 3.24. Mollusc biodiversity based on taxa richness (S) for reserve (R) and non-reserve (NR) sites surveyed in 2009. Refer to CD-Rom for additional biodiversity measures.

Cookia sulcata was numerically abundant across sites occurring at highest densities within the reserve at R1, R6 and R7, and was < 5 m^{-2} at the remaining reserve sites (Fig. 3.25). *Cookia sulcata* was generally more-abundant at sites outside of the reserve, attaining high to moderate densities > 5 m^{-2} at the majority of sites surveyed. Size frequency distributions for both reserve and non-reserve locations were right skewed, with individuals between 10 and 20 mm the most common and larger individuals between 40 and 70 mm much less common. Remnants of large *Cookia sulcata* were found at R2 and R3 adjacent lobster nests, and this species is a common food source for both lobster and stingrays (personal observation).

Trochus viridis was highly variable across reserve and non-reserve sites surveyed in 2009 (Fig. 3.25). Within the reserve highest abundances occurred in shallow-water locations R2 and R3, but were generally ≤ 2 m^{-2} at the other sites surveyed. For non-reserve sites there was no clear pattern with depth however *Trochus viridis* density was high > 8 m^{-2} for both NR7 and NR10, being ≤ 2 m^{-2} for the other non-reserve sites. Size frequency data was broadly similar between reserve and non-reserve areas although reserve sites had a higher frequency of individuals ≤ 10 mm, whereas non-reserve locations had a higher frequency of individuals > 10 mm.

Cantharidus purpureus attained highest densities (> 5 m^{-2}) within the reserve at deep-water sites (R4, R8, and R9) where *Ecklonia radiata* was abundant (Fig. 3.26). For non-reserve sites, with the exception of site NR2, low densities (< 1 m^{-2}) occurred across sites. The granose turban *Modelia granosa* exhibited maximum abundance in accordance with depth occurring in highest densities at reserve deep-water sites (R5, R8, R9, R10) and at deep-water non-reserve sites (NR2, NR9, NR10) (Fig. 3.26).

The paua *Haliotis iris* was more-common across shallow non-reserve sites occurring at highest densities at site NR1, but also occurring at sites NR4, NR7 and NR8 (Fig. 3.27). This taxa was only found within the within the reserve at site R3 (Gemstone Bay). Size frequency distribution data suggest that all sample populations were well under the legal size limit of 125 mm for this species.

Mixed model statistical analysis (GLIMMIX; Table 3.7) indicated that only *Cantharidus purpureus* was statistically different for the factor Status, being generally higher in the reserve across surveys, however abundance was not statistically different for the factor Year for any of the main taxa and Status ×Year interactions were also not statistically significant. The random effect in each model Site(Status×Year) was statistically significant for each taxa, due to the high variability at a site-specific level between surveys.

Table 3.7. Results from mixed model analysis (GLIMMIX) for selected gastropod (abundance within Hahei reserve and non-reserve sites between 2006 and 2009. Significance: $*p < 0.05$, $**p < 0.01$, $***p < 0.001$. Note: all Status×Year effects were non-significant.

Factor	**Fixed effect**		**Covariance parameter estimate**
	Status	**Year**	**Site (Status×Year)**
Cookia sulcata	$F_{1, 18} = 2.50$	$F_{2, 36} = 6.78$	Z value = 3.48, Pr $Z < 0.0001$
Trochus viridis	$F_{1, 18} = 1.56$	$F_{2, 36} = 1.25$	Z value = 6.10, Pr $Z < 0.0001$
Cantharidus purpureus	$F_{1, 18} = 4.97^{*}$	$F_{2, 36} = 3.24$	Z value = 5.87, Pr $Z < 0.0001$
Modelia granosa	$F_{1, 18} = 2.12$	$F_{2, 36} = 1.67$	Z value = 6.24, Pr $Z < 0.0001$

Less common gastropods found during the 2009 survey included the beaded top shell Callistoma punctulatum, the tiger shell *Callistoma tigris*, the circular saw shell *Astraea heliotropium,* the octagonal murex *Muricopis octogonus*, *Cominella quoyana*, and the smooth slipper shell *Mariocrypta monoxyla*.

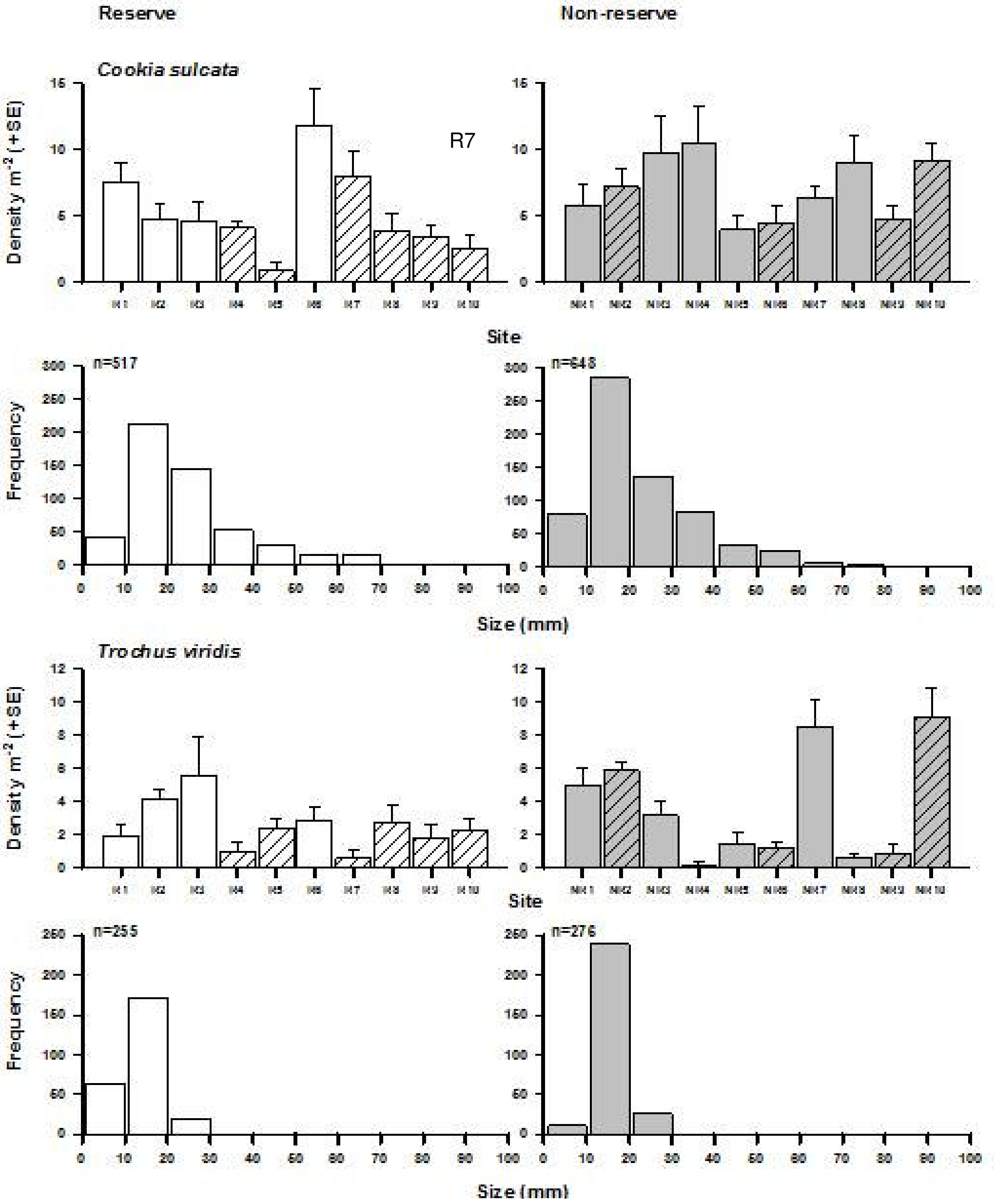

Figure 3.25. Density and size frequency distributions of *Cookia sulcata* and *Trochus viridis* within Te Whanganui-a-Hei marine reserve and non-reserve sampling sites in 2009. Open bars denote shallow-water sites (< 8 m depth) whereas hatched bars denote deep-water sites (> 10 m depth).

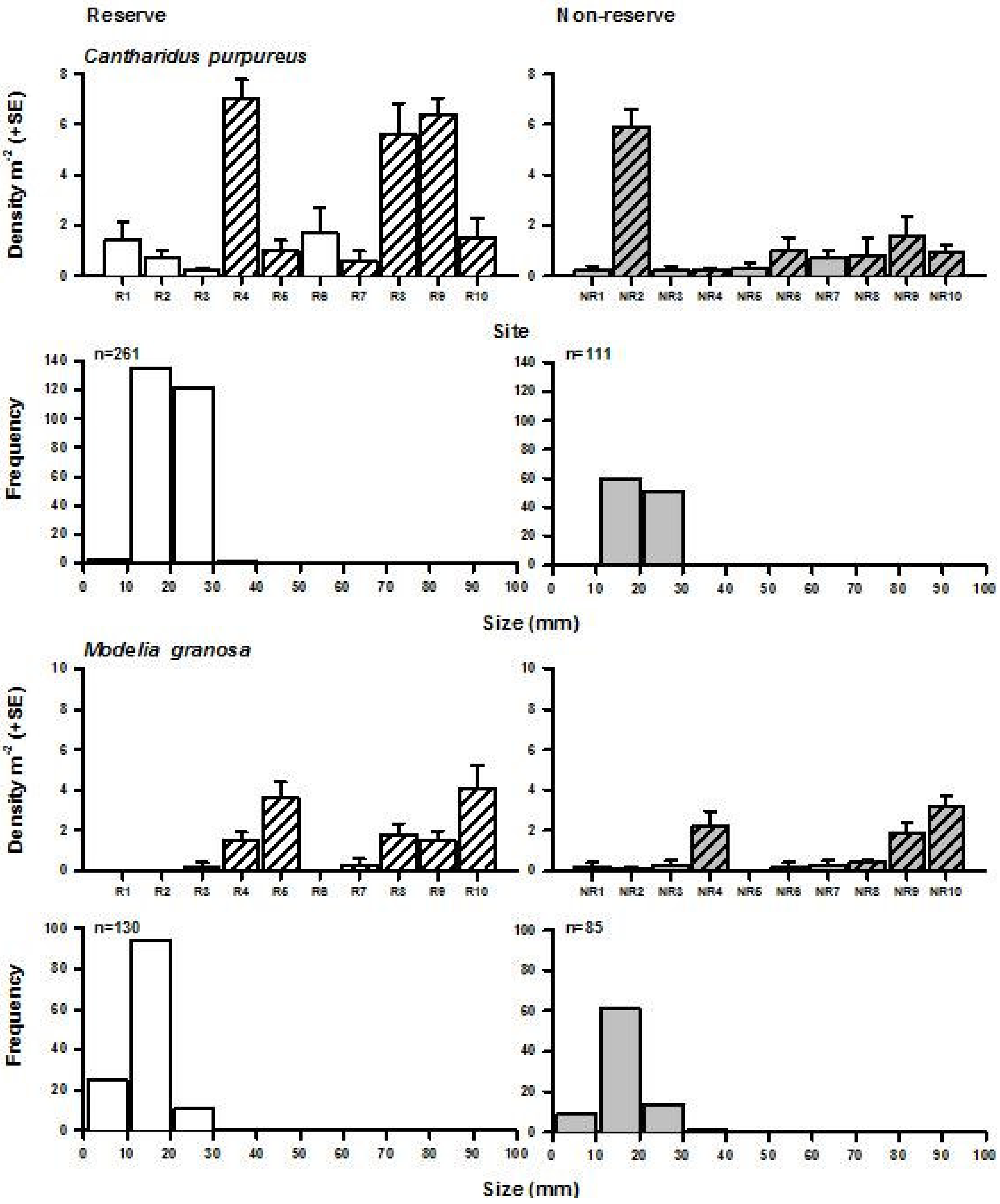

Figure 3.26. Density and size frequency distributions of *Cantharidus purpureus* and *Modelia granosa* within Te Whanganui-a-Hei marine reserve and non-reserve sampling sites in 2009. Open bars denote shallow-water sites (< 8 m depth) whereas hatched bars denote deep-water sites (> 10 m depth).

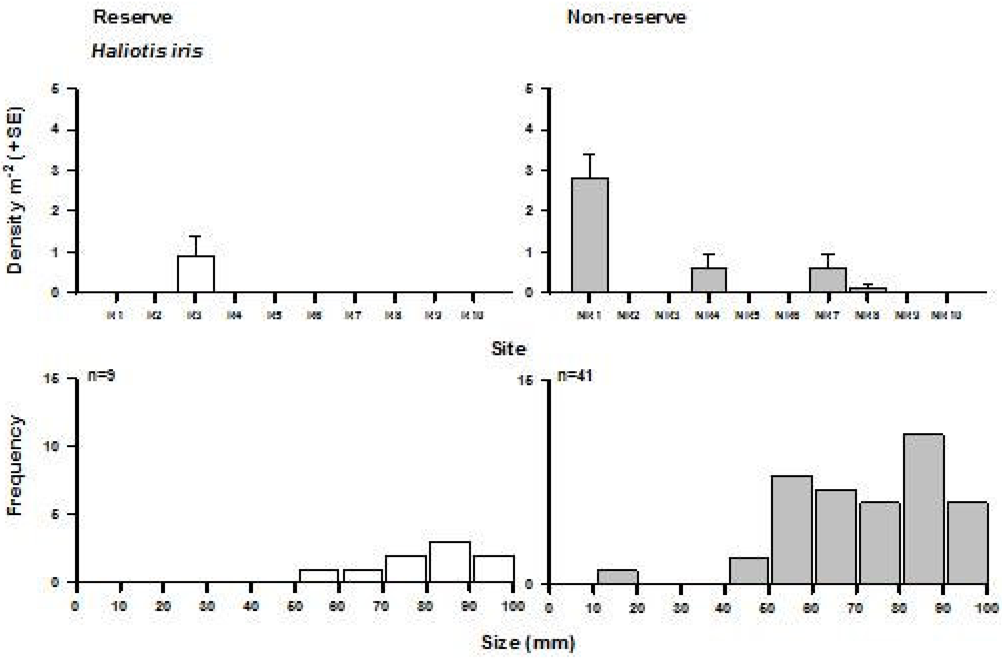

Figure 3.27. Density and size frequency distributions of *Haliotis iris* within Te Whanganui-a-Hei marine reserve and non-reserve sampling sites in 2009. Open bars denote shallow-water sites (< 8 m depth) whereas hatched bars denote deep-water sites (> 10 m depth).

3.4 Lobster

Abundance and size

Average abundance of *Jasus edwardsii* for all ten reserve site surveyed in 2009 was 20.30 lobsters per $500m^{-2}$ (+/- 4.2 – SE), which marks a slight decline in abundance from 2007 levels, yet remains substantially higher than 2006 levels, where lobster abundance was 12.11 lobsters per $500m^{-2}$ (+/- 3.3 – SE) (Fig. 3.11). As for previous years, there was high variability in lobster abundance across reserve sites surveyed (Fig 3.28). *Jasus edwardsii* were numerically dominant in shallow-water coastal sites within the reserve at sites R1 (Cooks Bluff), R2 (Boulder Bank), and R3 (Gemstone North), with abundances ranging between 35-60 lobsters per 500 m^2 (Fig. 3.29). In contrast, *J. edwardsii* abundance at deep-water and island sites was a great deal lower and typically < 10 lobsters per m^2 (Fig. 3.29). Relative to the last survey in 2007, lobster abundance declined at R2, R3, R4, and R7 and either increased or remained unchanged at the other sites (Fig. 3.29). Within shallow-water sites, larger lobsters were commonly gregarious and often found cohabiting dens, whereas lobsters were often solitary at deeper locations sampled. Of particular note was the behaviour of large males at sites R1 and R2, which were routinely observed in the open away from shelters and were particularly aggressive towards surveyors.

Jasus edwardsii size distribution (pooled across all reserve sites) within the reserve sample population in 2009 ranged from 30 mm to 230 mm C.L., with lobsters between 90 and 130 mm C.L. having the highest frequencies (Fig. 3.30). This demonstrates clear growth of the sample population since 2006 when lobsters between 80-90 mm had the highest frequency (Fig. 3.13). There has also been an increase in the frequency of larger lobsters > 180 mm C.L. over this period. Mean lobster size for the reserve sample population in 2009 was 114 mm ± 3.3 (95 % CI).

Lobster abundance in non-reserve areas in 2009 was 4.30 lobsters per $500m^{-2}$ (+/- 1.2 – SE), and the highest recorded since the change in sampling design in 2006. Within the non-reserve area surveyed, highest lobster densities occurred at NR4 and NR5, but there was no obvious trend in abundance associated with depth (Fig. 3.12). Lobster abundance increased at NR4, NR5, NR7, and NR10 between 2007 and 2009 and either declined or remained unchanged at the other sites surveyed. Site NR8 was of particular note due to high abundance of juvenile lobsters < 50 mm C.L., which were also apparent in the 2007 survey. As for previous surveys, the non-reserve sample population in 2009 was typified by very few legal-sized lobsters, with the majority of lobsters surveyed being less than 100 mm CL (Fig. 3.13). Mean lobster size for the non-reserve sample population in 2009 was 76.6 mm ± 4.8 (95 % CI).

The abundance of *Jasus edwardsii* (pooled across sites and depths) within the Te Whanganui-a-Hei Marine Reserve in 2009 was approximately 5 times higher than non-reserve areas surveyed (Fig. 3.11). Statistical analysis of 2009 abundance data (using mixed model analysis – GLIMMIX, SAS (1999)) indicated statistically significant differences in abundance between reserve and non-reserve areas (Status) (Table 3.8). The random effect in the model Site (Status) was also statistically significant indicative of the high heterogeneity among sites within reserve and non-reserve areas.

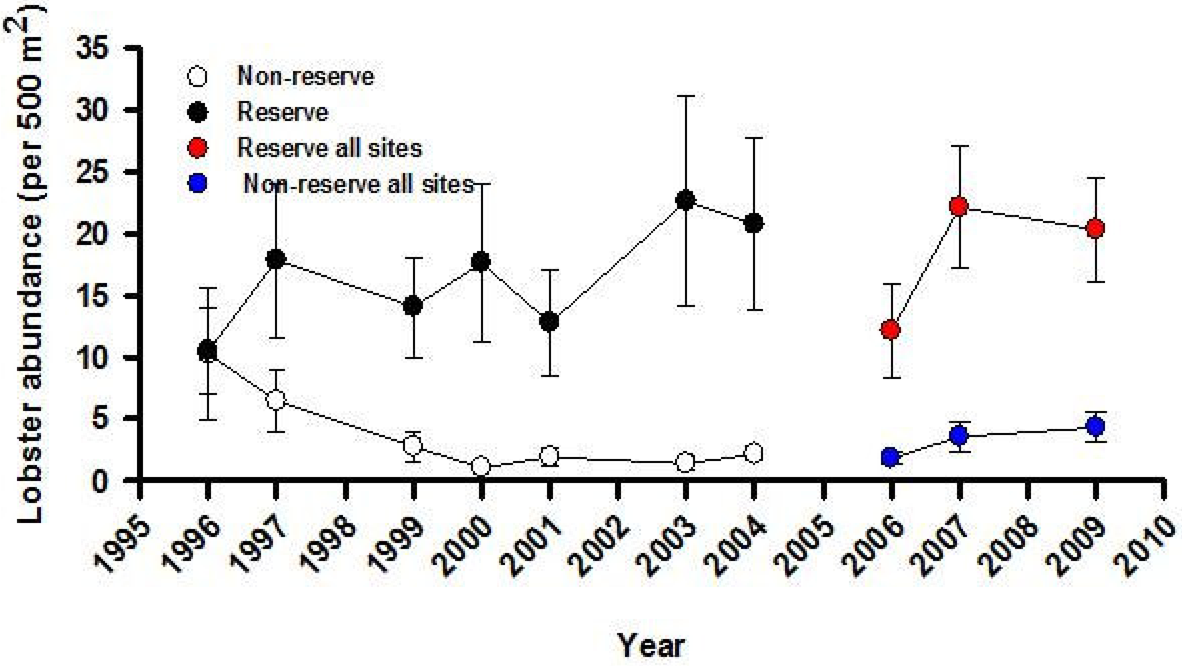

Figure 3.28. Mean abundance of *Jasus edwardsii* (± SE) pooled from sites within and outside Te Whanganui-a-Hei Marine Reserve from 1996 to 2009. Note in 2006 the sampling design was changed to include increased sites within the reserve and non-reserve areas.

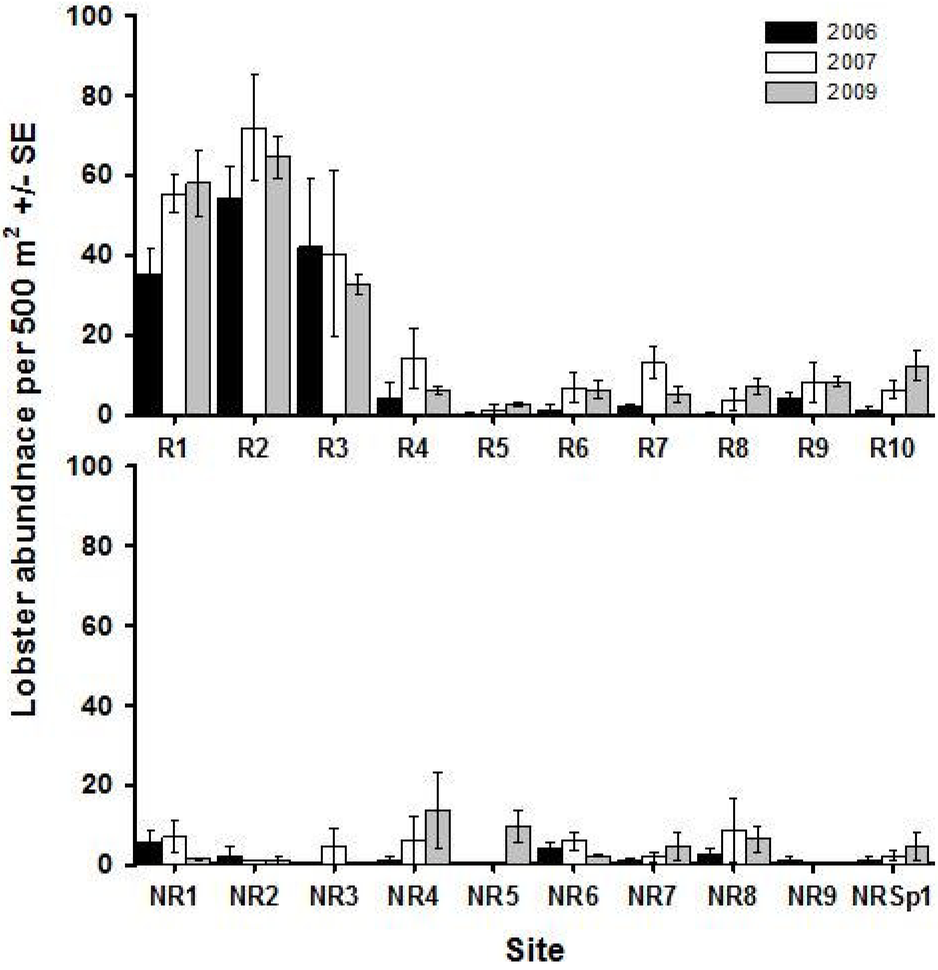

Figure 3.29. Mean abundance of *Jasus edwardsii* (± SE) within Te Whanganui-a-Hei Marine Reserve and non-reserve sites in 2006, 2007 and 2009. Refer to Fig. 2.1 for the location of all sites.

Analysis of the temporal dataset (2006-2009) indicated a statistically significant difference for the factor Status, although the factor Year was not statistically significant (Table 3.9). We interpret the statistically significant random effect in the model Site(Year×Status) to be indicative of lobster abundance changing at different rates among sites within reserve and non-reserve sample populations through time, e.g., between 2006 and 2007 lobster abundance almost doubled in the reserve population, although not across all sites, whereas the rate of change in the non-reserve population was comparatively lower. Estimates of the effect size, based on relative odds ratios between reserve and non-reserve areas indicates that lobster abundance between 2006 and 2009 was 4.93 (1.7, 14.2 95% CI) times higher within the reserve. In 2009, mean lobster size was statistically different between reserve and non-reserve areas (Two sample t-test, $P < 0.001$).

Table 3.8 Results from mixed model analysis (GLIMMIX) for lobster (*Jasus edwardsii*) abundance within Hahei reserve and non-reserve sites in 2009. Significance: *p < 0.05, **p < 0.01, ***p < 0.001.

Factor	Fixed effect	Covariance parameter estimate
Jasus edwardsii	**Status**	**Site (Status)**
	$F_{1, 18} = 8.61$***	Z value = 1.65, Pr $Z = 0.045$

Table 3.9 Results from mixed model analysis (GLIMMIX) for lobster (*Jasus edwardsii*) abundance within Hahei reserve and non-reserve sites between 2006 and 2009. Significance: *p < 0.05, **p < 0.01, ***p < 0.001. Note: the Status×Year effect was non-significant.

Factor	Fixed effect		Covariance parameter estimate
Jasus edwardsii	**Status**	**Year**	**Site (Status×Year)**
	$F_{1, 56} = 8.73$**	$F_{2, 56} = 2.00$	Z value = 4.10, Pr $Z < 0.0001$

Lobster and urchin abundance

In 2006 bivariate plots of exposed *Evechinus chloroticus* abundance relative to *Jasus edwardsii* abundance suggested possible site-specific predatory effects where the both co-occurred in shallow-water (Fig. 3.31). This pattern held true for a range of sites in 2009, whereby shallow-water reserve sites with high lobster abundance tended to have a lower mean density of exposed urchins compared to non-reserve shallow-water sites.

This pattern was less-apparent for *Evechinus chloroticus* size in 2006 (Fig. 3.31), but in 2009 there was almost a linear trend with respect to the mean size of exposed *Evechinus chloroticus* in relation to *Jasus edwardsii* density for reserve sites; implying that where *Jasus edwardsii* abundance was high, exposed urchin size was also high. This pattern was not as apparent for non-reserve sites (Fig. 3.31).

Habitat variation

The current survey was the first to quantify habitat coverage across individual lobster transects at a site-specific level. A total of seven habitat types according to Shears *et al.* (2004) were enumerated across transects (Fig. 3.32) and general habitat structure showed accordance with depth. This was reflected by MDS ordinations which indicated no clear grouping of sites for the factor Status, however there was a distinct separation between shallow and deep-water sites (i.e., for the factor Depth) (Fig. 3.33) with shallow-water sites grouped to the left of the ordination, and the majority of deep-water sites grouped to the center of the ordination. Mixed algal stands and *Ecklonia radiata* patches were typical habitats of reserve shallow-water sites, and these habitats were also common within non-reserve shallow-water sites, in tandem with urchin barrens habitat. *Ecklonia radiata* habitat dominated all deep-water sites irrespective of status, with the exception of sites R5 and R10, clustered to the right of the ordination, where sponge flats habitat was predominant (Fig. 3.32).

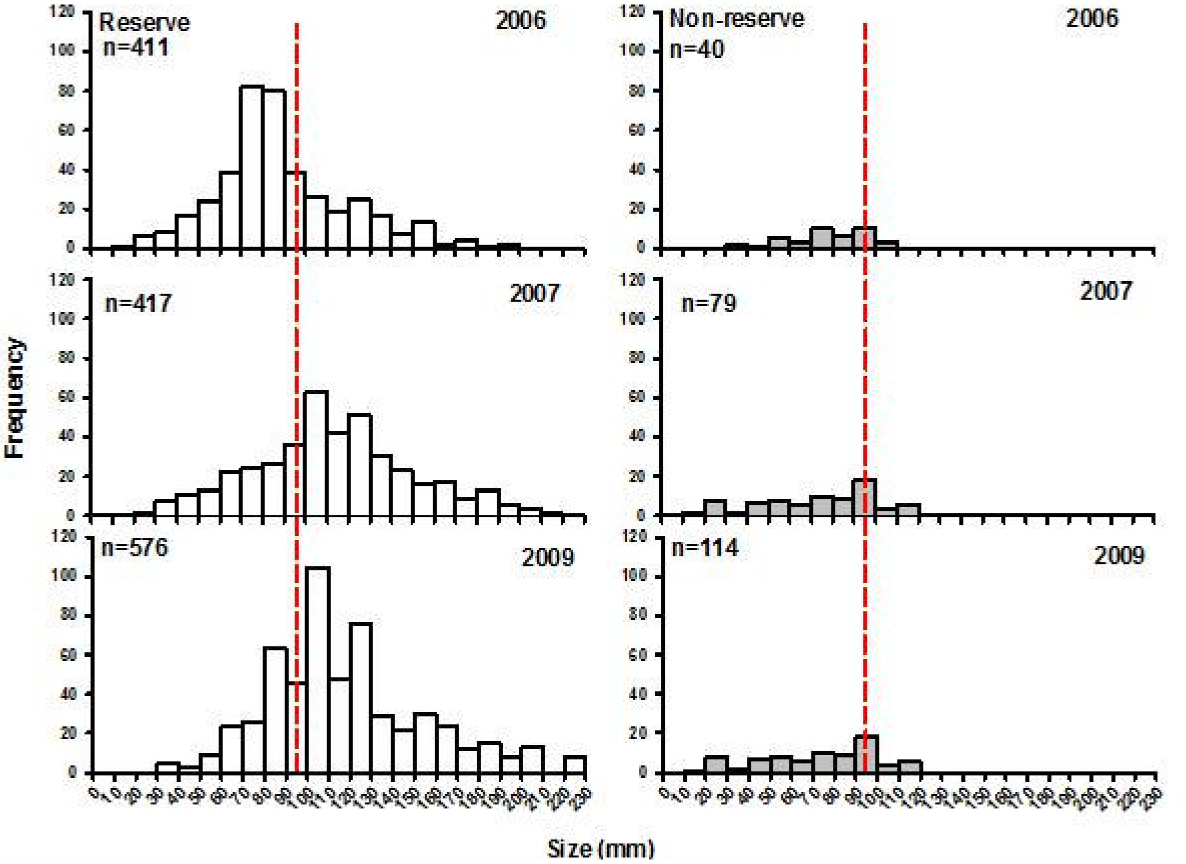

Figure 3.30. Size frequency histograms of *Jasus edwardsii* within Te Whanganui-a-Hei Marine Reserve and non-reserve control sites in 2006, 2007 and 2009. The dashed line denotes the division between legal and sub-legal lobsters

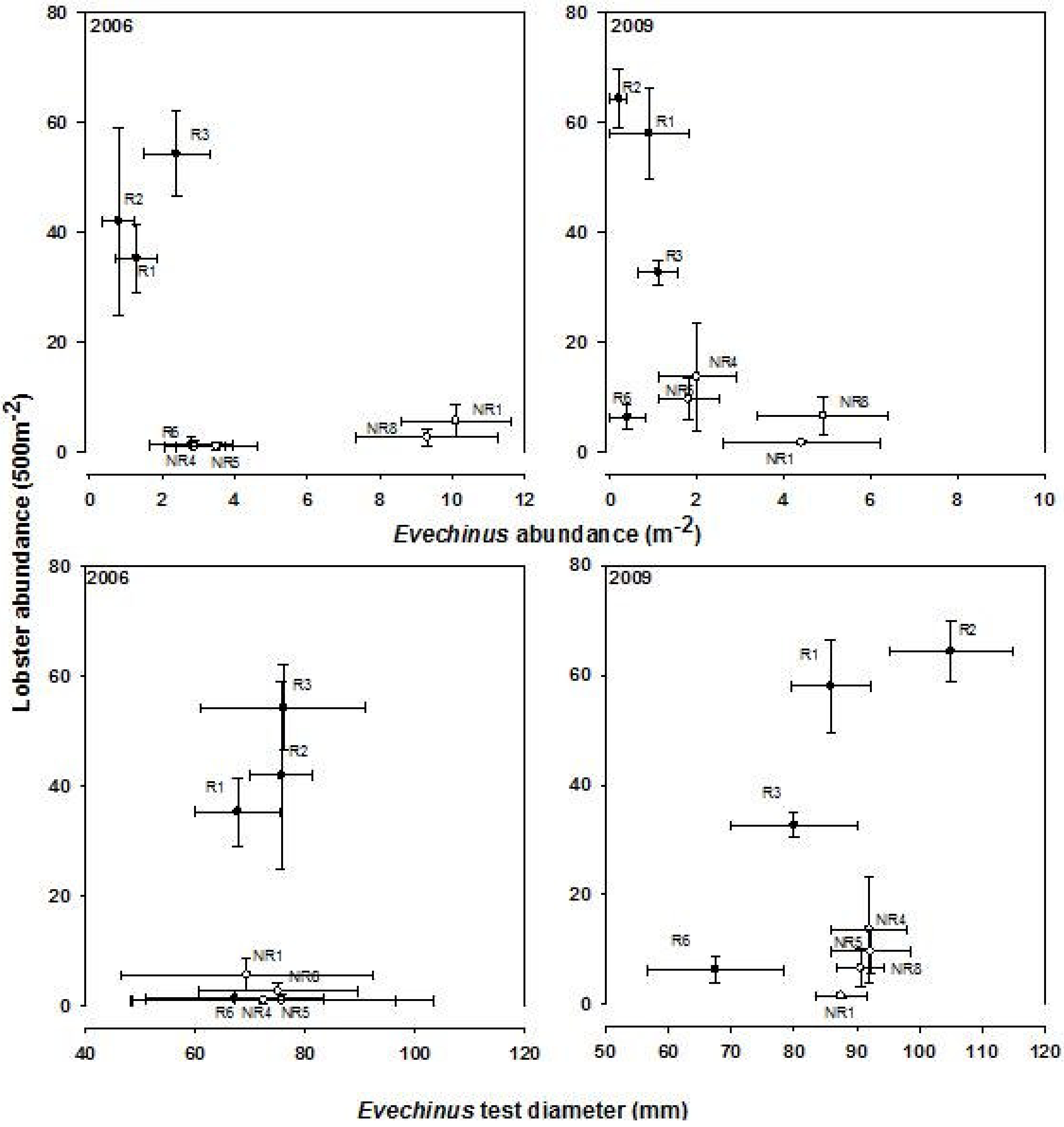

Figure 3.31. Bivariate plots of *Evechinus chloroticus* density (± SE) and size (± 95% C.I.) in relation to *Jasus edwardsii* density ((± SE) within Te Whanganui-a-Hei marine reserve and non-reserve sites in 2009.

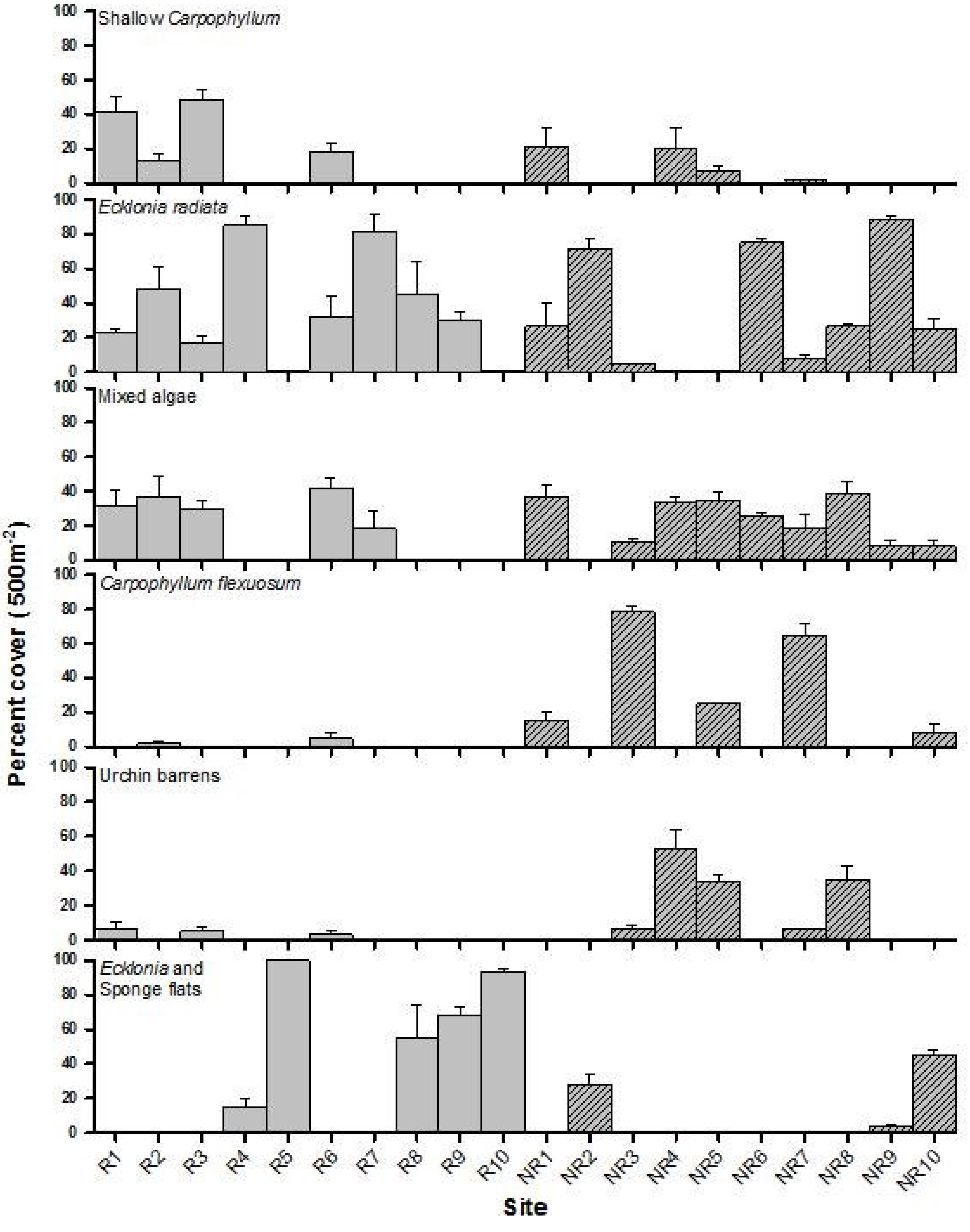

Figure 3.32. Mean percent cover of main habitats (see Shears *et al*. 2005 for habitat descriptions) across lobster transects within Te Whanganui-a-Hei marine reserve and non-reserve sites (hatched bars) in 2009.

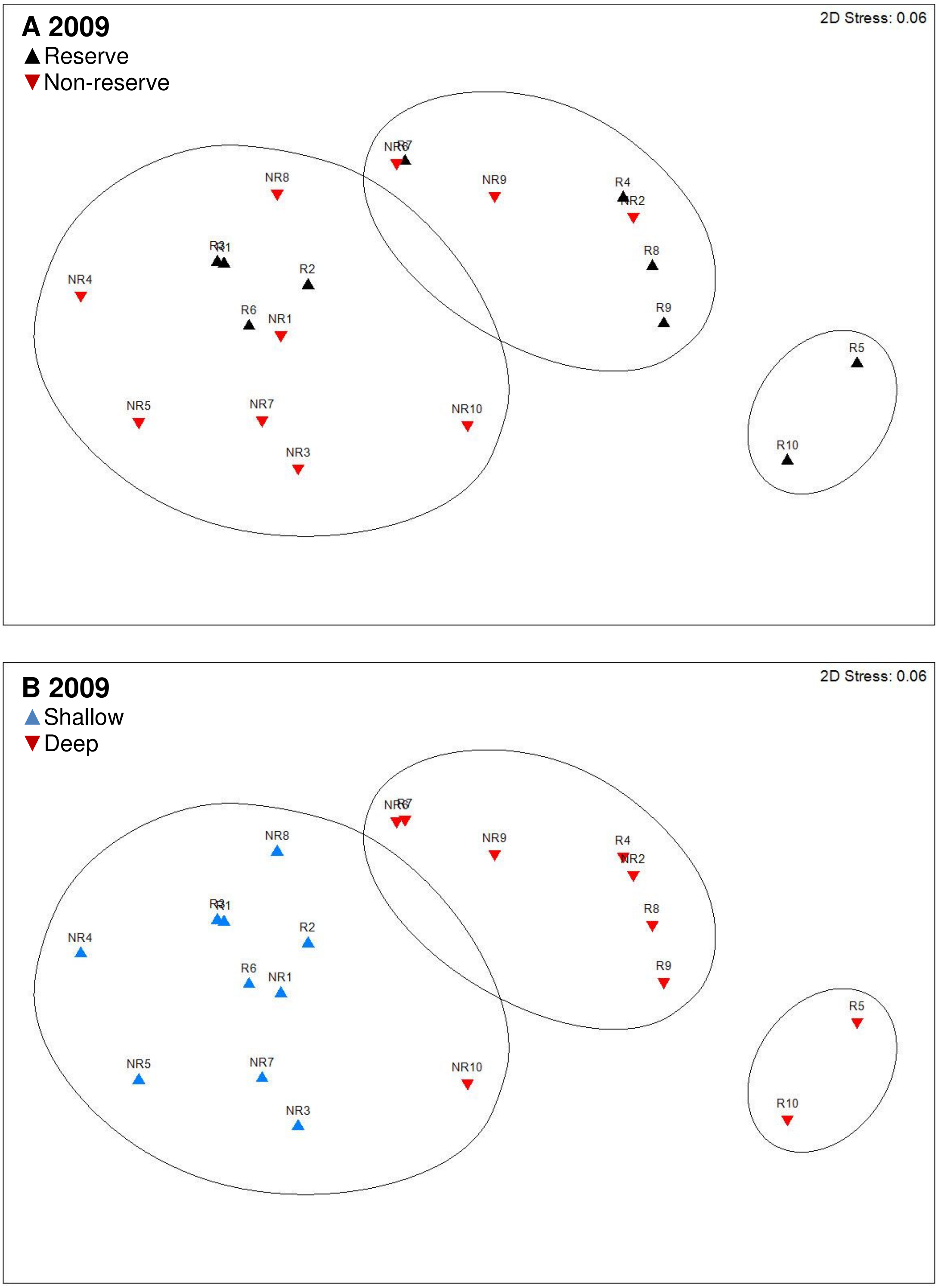

Figure 3.33. MDS ordinations for 2009 based on the percent cover of habitats across sites for A the factor status (reserve and non-reserve) and for B the factors shallow and deep. Clusters are depicted at the 50 % similarity level.

4.0 Discussion

This report details the second survey of $100m^2$ permanent plots within the Te Whanganui-a-Hei (Hahei) Marine Reserve and equivalent non-reserve areas since their establishment in 2006 together with lobster monitoring, which has been carried out since 1996, but since modified in 2006. The main objective of the biological monitoring of these subtidal rocky reef areas is for long-term trend detection capable of identifying threats to the reserve biota over which DOC can exert some management control, or which have the potential to cause changes to particular habitats, species and/or sections of the marine community. The subtidal rocky reef habitats quantified in this current study bode well in terms of the monitoring programme fulfilling its long-term objectives. Consecutive surveys of this nature enable the identification of changes in species abundance/distributions and evaluation of habitat stability and community biodiversity across reserve and non-reserve areas through time space and time.

The general focus behind the survey methodology was to stratify the sampling based on habitat types as depicted from a habitat map of the marine reserve created in 2004 (NIWA 2004). Unfortunately, in the majority of instances, sites that have been physically sampled within the reserve do not closely match those habitats depicted on the map and equivalent non-reserve rocky reef sites continue to be difficult and/or impossible to find. This is, in part, due to the complex array of habitats within the reserve relative to outside, but more-importantly due to the lack of a comparable non-reserve habitat map. In addition, the three rocky reef habitat classification types used to construct the map were somewhat generic in relation to other studies focused on quantifying and describing subtidal rocky reef habitats in northern New Zealand (see Ayling 1981; Choat and Schiel 1982; Shears *et al.* 2004), including habitat descriptions used in previous surveys of subtidal rocky reef communities within Hahei (Shears *et al.* 2000; Haggitt and Mead 2007). Of particular note, missing from the habitat map is the shallow *Carpophyllum* and *Ecklonia* habitat that has been present between Cooks Bluff and Gemstone Bay for at least the last 13 years (personal observation), as well as urchin barrens habitat, which is patchily distributed within the reserve on the northern side of Motueka Island (Site RR6) and between Gemstone Bay and Mahurangi Island (personal observation in 2006; also see Shears *et al.* 2000). It is our view that habitat maps of this nature need to depict all significant habitats, so that habitat stability/change can be examined at a functional level, particularly when used to underpin site selection for monitoring programmes.

Biological Monitoring - General Findings

The range of subtidal rocky reef habitats and biological communities enumerated in 2006 provided a structured baseline for biological sampling in 2009. Within the reserve, rocky-reef sites were dominated by algal habitats that change consistently with depth (Choat and Schiel 1982; Shears *et al.* 2004) and to a lesser degree exposure (Choat and Schiel 1982; Shears *et al.* 2000, 2004). While patterns in density and biomass were generally comparable to 2006 levels, two differences were apparent in 2009. Firstly, *Ecklonia radiata* biomass at several deep-water sites was notably lower than 2006, and secondly, *Carpophyllum maschalocarpum* density and biomass had increased substantially at mainland shallow-water sites.

The reduction in *Ecklonia radiata* biomass was attributable to canopy thinning and patchy dieback. Studies of *Ecklonia radiata* dieback within north-eastern New Zealand suggest that it is a natural process, and generally stands recover to former densities (Haggitt and Babcock 2003), although recovery is dependent on demographic attributes of *Ecklonia radiata*, particularly the season of canopy regression, and the agent(s) of mortality (e.g., light attenuation, amphipod grazing, viruses). The increase of *Carpophyllum maschalocarpum* at shallow water sites (R1, R2 and R3) reflects demographic changes in this alga through space and time and variable nature of shallow water macroalgal populations in general (Anderson *et al*. 2005; Haggitt and Mead 2007). Large changes in the biomass of macroalgal taxa among biological surveys are reasonably typical, for example Shears (2007) documented a doubling of *Ecklonia radiata* biomass at Leigh, Poor Knights Island and Mokohinau Island survey sites between 1999 and 2006 reflective of regional-wide variation over this period. Subsequent surveys will track *Ecklonia radiata* recovery and additional changes to reserve macroalgal assemblages, which continues to be an important directive.

Non-reserve habitats within permanent plots initially surveyed in 2006 also conformed to typical rocky reef habitat structure in north-eastern New Zealand (Choat and Schiel 1982; Shears and Babcock 2003), with urchin barrens habitat interdispersed with small patches of mixed algal habitat in shallow-water and *Ecklonia radiata* forest common below 10 m depth (Haggitt and Mead 2006). While *Ecklonia radiata* is still abundant in non-reserve deep-water sites, one of the most prominent observations for shallow-water non-reserve sites in 2009 was the increase in density and biomass of *Ecklonia radiata* and fucalean taxa, together with a general decline in the density of *Evechinus chloroticus*. At many of these sites (based on habitat information from lobster transects) increased macroalgal cover extended well beyond the permanent plots indicating a general reduction in barrens habitat (personal observation).

Despite the increase in macroalgal diversity and biomass in these shallow-water sites in 2009, macroalgal biomass still remains higher within the reserve, consistent with the trophic cascade paradigm evident in other marine protected areas in Australasia (Shears and Babcock 2003; Barrett *et al*. 2008). In their survey of biological populations within Te Whanganui-a-Hei Marine Reserve, Shears *et al.* (2000) acknowledged that reserve-related changes were possibly manifest in 2000, seven years following the establishment of the reserve. These authors documented significant differences between reserve and non-reserve sites with regard to the density of *Ecklonia radiata*, fucalean algae and *Evechinus chloroticus*, with *E. chloroticus* population structure in the reserve characterised by smaller urchins and a higher proportion of cryptic individuals, pattern analogous to the present study. Future surveys will establish if macroalgal biomass continues to increase further at non-reserve sites, and whether such changes result in long-term habitat shifts.

Increased macroalgal biomass in shallow-water non-reserve sites between 2006 and 2009 may be due to the reduction in *Evechinus chloroticus* recorded over this period and changes of this nature are not uncommon on temperate rocky reefs elsewhere (Ebeling *et al*. 1985; Edgar and Barrett 1997; Scheibling and Hennigar 1997; Shears and Babcock 2003; Barrett *et al*. 2008). Factors that favour algal recruitment (see Harrold and Reed 1985) and macroalgal growth may also be paramount and match those occurring around the Leigh coastline where, in particular, *Carpophyllum flexuosum* has increased in density and biomass on wave exposed shores (Cole *et al*. 2001). Mechanisms responsible for this distributional expansion appear complex, but may be related to decreased wave energy (on a decadal scale), its low palatability (Cole and Haggitt 2002) or a combination of these factors.

Due to the ability of *Evechinus chloroticus* to directly influence algal abundance, perhaps the most important factor is to determine what mechanisms are likely to be affecting this species.

Past surveys in the Hahei area have measured and anecdotally observed moribund urchins in barrens habitat, characterised by folded guard spines, and spine loss, which may lead to death (N. Shears personal communication). Under the current programme *E. chloroticus* with spine loss were only observed at NR4 in 2006, which is now characterised by mixed algal stands and urchin barrens. Due to the infrequency of sampling and the patchy nature of events through space and time, it is conceivable that these events are missed. Lafferty (2004) noted the importance of disease in reducing urchin populations and thus indirectly influencing macroalgal habitat structure in the Channel Islands National Park, California, although suggested that disease may not affect urchin abundance as greatly as natural predation.

The 2009 survey also recorded a higher frequency of cryptic individuals in non-reserve areas, implying reduced urchin grazing. Studies suggest that such behaviour may be a direct response to increased predator densities such as snapper and lobster (Shears and Babcock 2003), but may also be related to increased agitation associated with increased macroalgal cover (Cole and Haggitt 2001).

Fish surveys conducted within Te Whanganui-a-Hei reserve and non-reserve areas have continually documented higher snapper abundance in reserve areas, suggesting that predation rates on *Evechinus chloroticus* may in-effect be higher in the reserve, and thus, coupled with lobster predation, negatively affect urchin abundance. The most recent survey (Taylor *et al.* 2006) noted a substantial increase in the numbers of legal-sized fish (> 270 mm fork length) outside the reserve, observed mainly in the two southeast reference areas. This increase resulted in the lowest reserve:non-reserve ratio of legal-sized snapper and suggests that predation may also be increasing in non-reserve areas and large snapper were observed at non-reserve sites NR4 and NR9 during biological monitoring. Reef-fish monitoring will occur in 2010 and will evaluate whether the patterns identified by Taylor *et al.* (2006) are still apparent.

In recent years, several species have been identified as potentially problematic with regard to species biodiversity in temperate coastal waters in New Zealand, the most notable of these being the brown alga *Undaria pinnatifida* (Russell *et al.* 2008) and the clubbed tunicate *Styela clava* (biosecurity New Zealand). While these species have the potential to occur within Te Whanganui-a-Hei reserve and surrounding coastline, neither species was observed in 2009. There was no evidence of other invasive species regarded as problematic such as *Caulerpa taxifolia* and the paddle-crab *Charybdis japonica*.

Previous studies done within Te Whanganui-a-Hei Marine Reserve have recognised (Shears *et al.* 2000) and, in part, quantified (Schwarz *et al.* 2004; Steger 2006) the physical gradient that exists from western to eastern areas of the reserve – specifically wave exposure, water clarity and sedimentation variables. Shears *et al.* (2000) found habitats to vary in accordance with these abiotic factors however, the study of Schwarz *et al.* (2004) could not detect associations in understorey organisms (counts or percentage cover), consistent with a sedimentation gradient. Steger (2006) provides a well-structured study indicating a very real, but variable gradient from west to east across the reserve. In that study, data indicated that increased suspended and deposited sediment were an important part of the disturbance regime, but was variable in effects over space and time adding to the biological variation within the reserve. Such effects were also predicted to be greater in La Niña climatic periods when rainfall in north-eastern New Zealand is generally higher.

Consistent with past studies (Shears *et al.* 2000; Haggitt and Mead 2006) in 2009 there was a high percent cover of fine sediment on the survey reefs within northern shallow-water and deep-water sites within the reserve, with sediment levels much lower at non-reserve sites south of the marine reserve. The higher average sediment levels in 2009 compared to 2006 within many of the deep-water reserve sites surveyed may be directly related to lower biomass and cover of *Ecklonia radiata*, as macroalgae can trap and redistribute fine sediment, reducing its cover on the substratum (Airoldi 2003; Steneck *et al.* 2002; Graham 2004). This does not however explain increases in sediment at shallow-water reserve sites such as R1 and R2, although this may be related to lower exposure levels and the proximity to Whitianga Harbour and Purangi Estuary relative to other shallow-water sites surveyed. Regardless, the present-day sediment levels indicate a sustained impact likely derived from these two catchments (Schwartz *et al.* 2004; Steger 2006). Increased levels of fine sediment on the substratum are of concern, particularly with regard to reduced biodiversity and community functioning (Airoldi 2001).

Because fine sediment tends to settle in deep-water where orbital wave velocities are lower than in shallow-water, deep-water sites within the Te Whanganui-a-Hei marine reserve have a greater potential to be adversely affected by sedimentation. In this study, sessile invertebrate communities at deep-water sites were most notable for their diverse sponge, ascidian, bryozoan and cnidarian communities that actively filter the water. These communities are particularly important to ecosystem functioning and must be considered special areas of biodiversity within the reserve. Sessile invertebrate taxa richness was relatively similar across sites between 2006 and 2009 surveys, suggesting little change has occurred between surveys, however the higher sediment levels in deep-water is of concern with regard to sessile invertebrate diversity and functioning. Experimental evaluations suggest that fine sediment can adversely affect the survival and diversity of these communities, particularly those that are encrusting in nature or low-lying. Steger (2006) also demonstrated the effects of sedimentation on select sessile invertebrates including the sponge *Tethya aurantium* and the mussel *Perna canaliculus* establishing that while effects may vary among taxa, all show adverse responses to higher turbidity and suspended sediment levels that include reduced growth and condition. However several studies suggest that sponges are more abundant and diverse in areas of high sedimentation due to reduced UV light as a result of increased turbidity (Roberts *et al.* 1998; Bell 2004). The degree to which sedimentation influences biological habitats within the Te Whanganui-a-Hei Marine Reserve continues to be of utmost importance and recurrent monitoring will ensure that this can be quantified relative to any changes in biological community structure and diversity.

Sand inundation on deep-reef areas in the center of the reserve in 2009 is also likely to be reflective of recent oceanographic processes related to La Niña oceanographic conditions that have been occurring between 2007 and 2009. La Niña events for this region of New Zealand are typified by increased wave action from the North-East and there have been notable large-scale wave events both in 2008 and 2009 likely to have lifted sand onto adjacent reef habitat. While sand inundation is a natural process and biological communities are likely to have evolved in response to such events, impacts to sessile invertebrate may include reduced diversity, and a shift from a relatively mature and stable community to a more unstable, less diverse community (Carballo 2006).

Lobster - General Findings

Lobster populations within Te-Whanganui a Hei Marine Reserve and adjacent non-reserve areas have been monitored at least biannually since 1996 (see Haggitt and Mead 2007). The continually higher lobster abundance (between 15–20 lobsters per 500 m^{-2}) within the reserve compared to sustained low abundance (< 5 lobsters per 500 m^{-2}) outside the reserve since 2000 demonstrates unequivocally the protective role of marine reserves, but also the negative effects of fishing (Shears *et al.* 2006: Barrett *et al.* 2008). Temporal data suggest that the reserve population has remained reasonably stable over the last 2-3 years, following a reduction in abundance of legal-sized lobsters within the reserve population between 2004 and 2006. Such a reduction may be indicative of fishing pressure around Hahei, as lobsters may move beyond the reserve boundary (behaviour associated with moulting, reproduction and feeding) where they come susceptible to fishing (see Kelly 2001). In a recent study Freeman (2008) suggests that where reserve boundaries traverse continuous reef habitat, movement beyond the reserve boundary is likely to occur. Applying this concept to Te-Whanganui a Hei Marine Reserve, *Jasus edwardsii* emigration is most-likely to occur along the offshore boundary north of Moturoa and Motueka Islands where deep-reef continues beyond the reserve boundary (NIWA 2004).

Surveys undertaken in 2007 and 2009 have demonstrated that a strong recruitment pulse evident in 2006 has now transpired into a high abundance of legal-sized lobsters between 100-130 mm C.L. within the reserve population. Recruitment pulses have also been evident in non-reserve areas, but these have not transpired into high abundances of legal-sized lobster. The absence of similar patterns in non-reserve areas suggests that the presence of adults could be important to juvenile emigration and survival, due to elevated protection afforded by adult conspecifics (Childress & Hernkind 1997; Butler *et al.* 1999). Fishing outside the reserve may also be having a greater impact than the immediate removal of legal-sized individuals, as Freeman and MacDiarmid (2009) demonstrated that sublegal *J. edwardsii* in the fishery are handled more frequently and suffer higher mortality rates as a consequence.

Jasus edwardsii abundance at several shallow non-reserve sites (NR4, NR5, and NR8) has increased since the 2007 survey and two of these sites (NR4 and NR5) have experienced increased macroalgal cover over this time period. Studies that have focused on habitat change in marine reserves have identified the significance of top-down processes with species such as snapper and lobster having the potential to control urchin populations (Shears and Babcock 2002, 2003; Barrett *et al.* 2008). Presently, relating habitat change within these non-reserve sites (and within the reserve) to the abundance of predatory organisms such as snapper and lobster is difficult, as changes outside of the reserve have only recently occurred.

Conclusion and Recommendations

In its present form, the Te Whanganui-a-Hei Marine Reserve biological monitoring programme is meeting its objectives but could be improved in several areas. These include:

1) Development of a detailed habitat map of the non-reserve survey area. This should go as far as to include a re-examination of the existing reserve habitat map, with habitats being defined and classified according to Shears *et al.* (2004) – we note that Hewitt *et al.* (2004) had some difficulty in using this classification system. Nevertheless,

considering the data collected in this survey and particularly given the recent habitat changes in shallow-water non-reserve sites, we feel that this is still an important and achievable directive. Following a re-analysis of habitat types, an assessment can be made on sampling design prior to the next survey;

2) Reducing the number of sites surveyed within the reserve and non-reserve areas to 8 per area (rather than 10). The principle reason for this is to ensure that a balanced sampling design for the factor depth is attained, i.e., 4 shallow and 4 deep sites per area rather than the unbalanced sampling design (for the factor depth) that presently exists. This will bode well for more in-depth and meaningful statistical analysis to be undertaken.
3) Collection of data on physical variables and/or undertaking analysis of physical variables already collected in other studies, (e.g., by Kate Steger) would be invaluable to assess the importance of abiotic factors (turbidity/sedimentation and exposure) in determining the structure of habitat types within the Hahei coastal area.

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Appendix 1.0

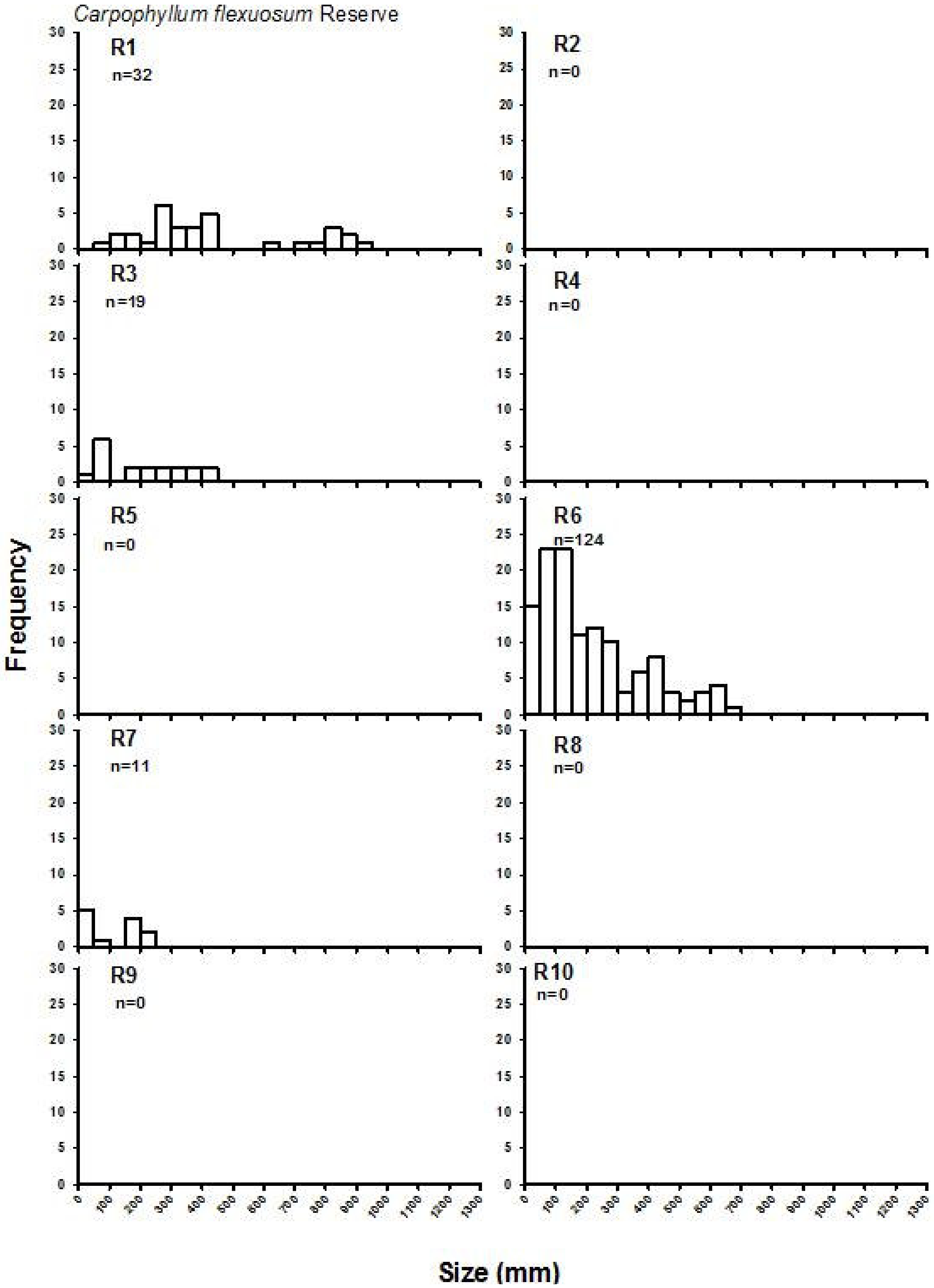

Figure A1. Size frequency distribution of *Carpophyllum flexuosum* within reserve sampling sites in 2006.

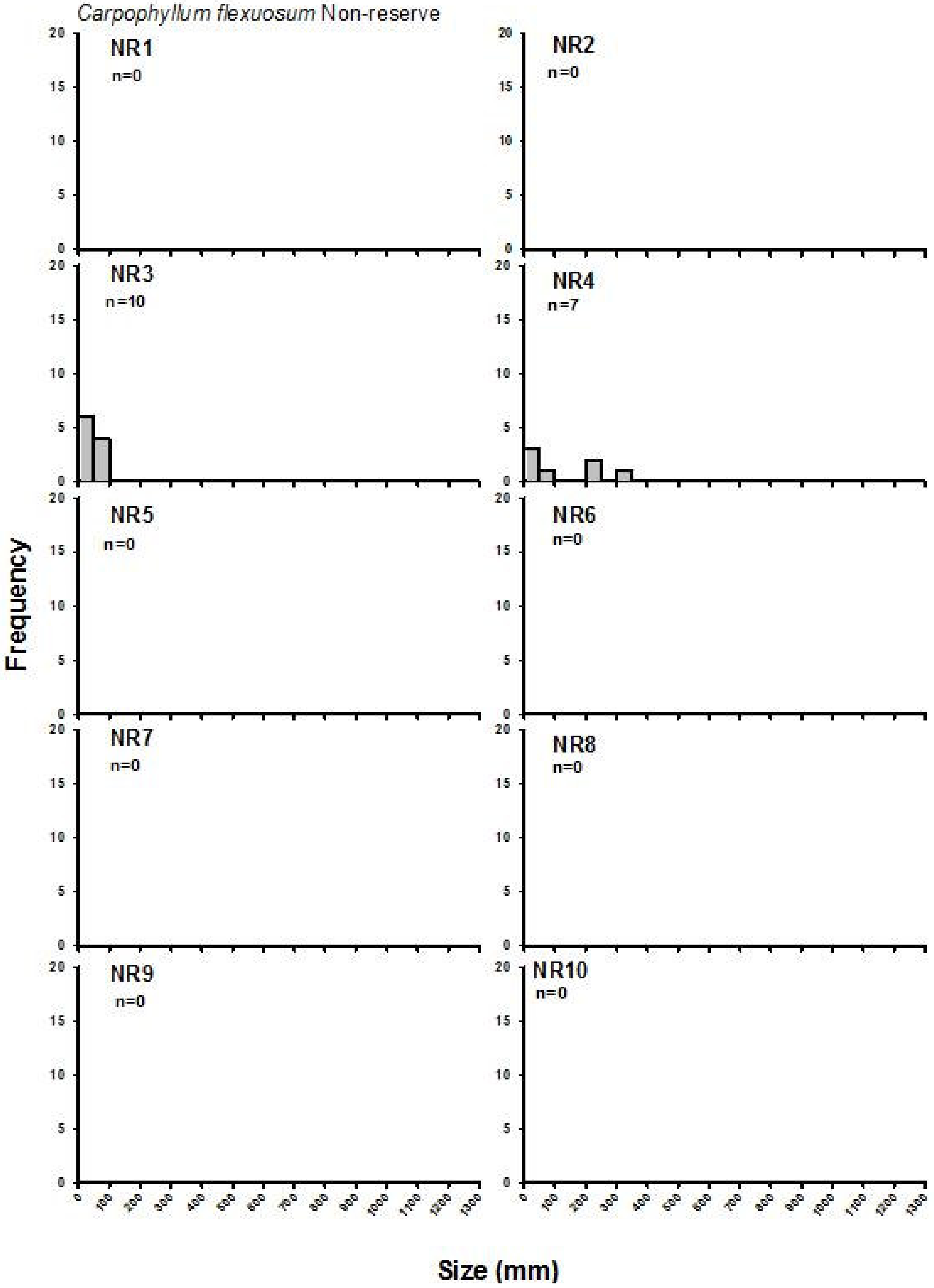

Figure A1 Continued. Size frequency distribution of *Carpophyllum flexuosum* within non-reserve sampling sites in 2006.

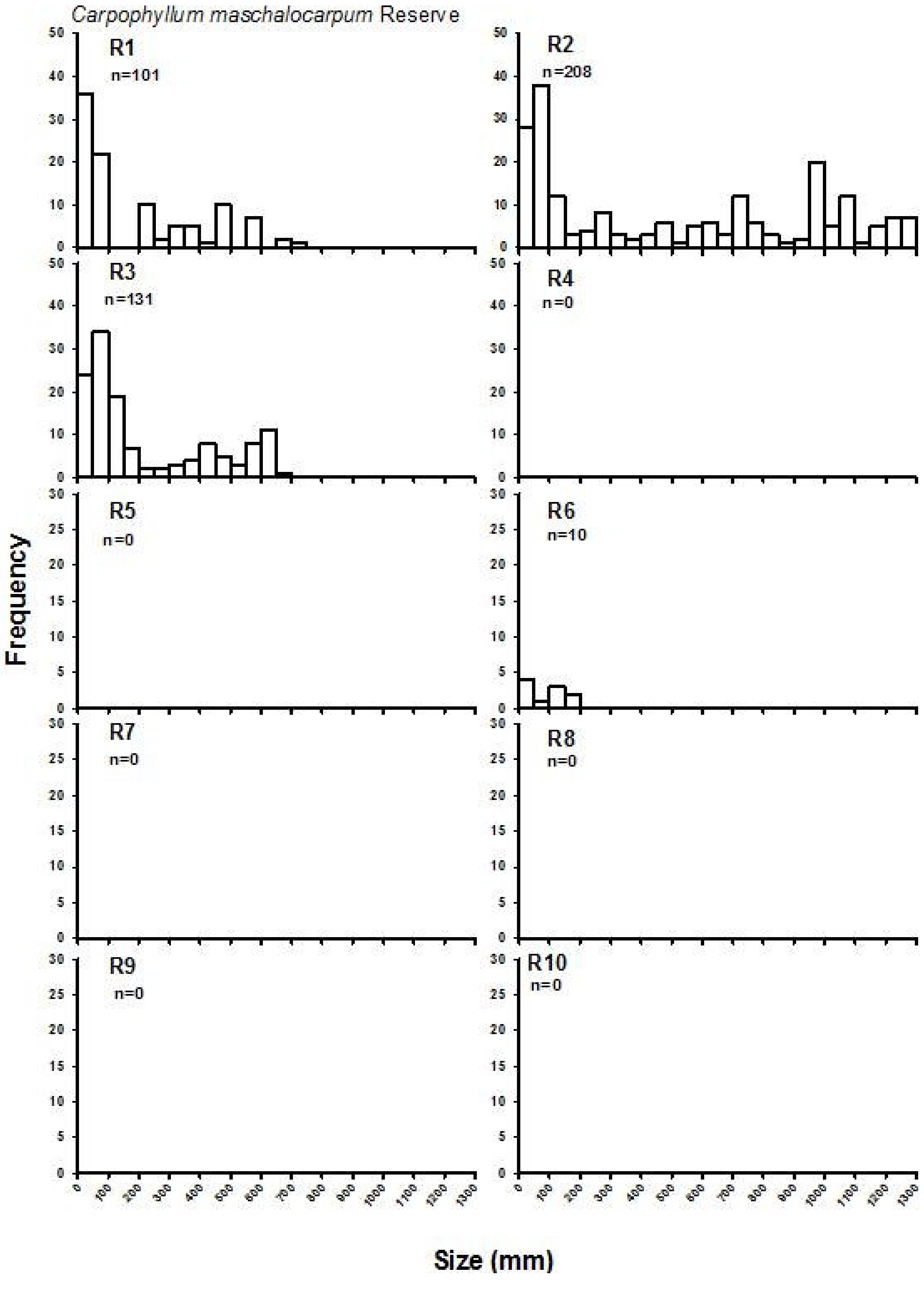

Figure A1 Continued. Size frequency distribution of *Carpophyllum maschalocarpum* within reserve sampling sites in 2006.

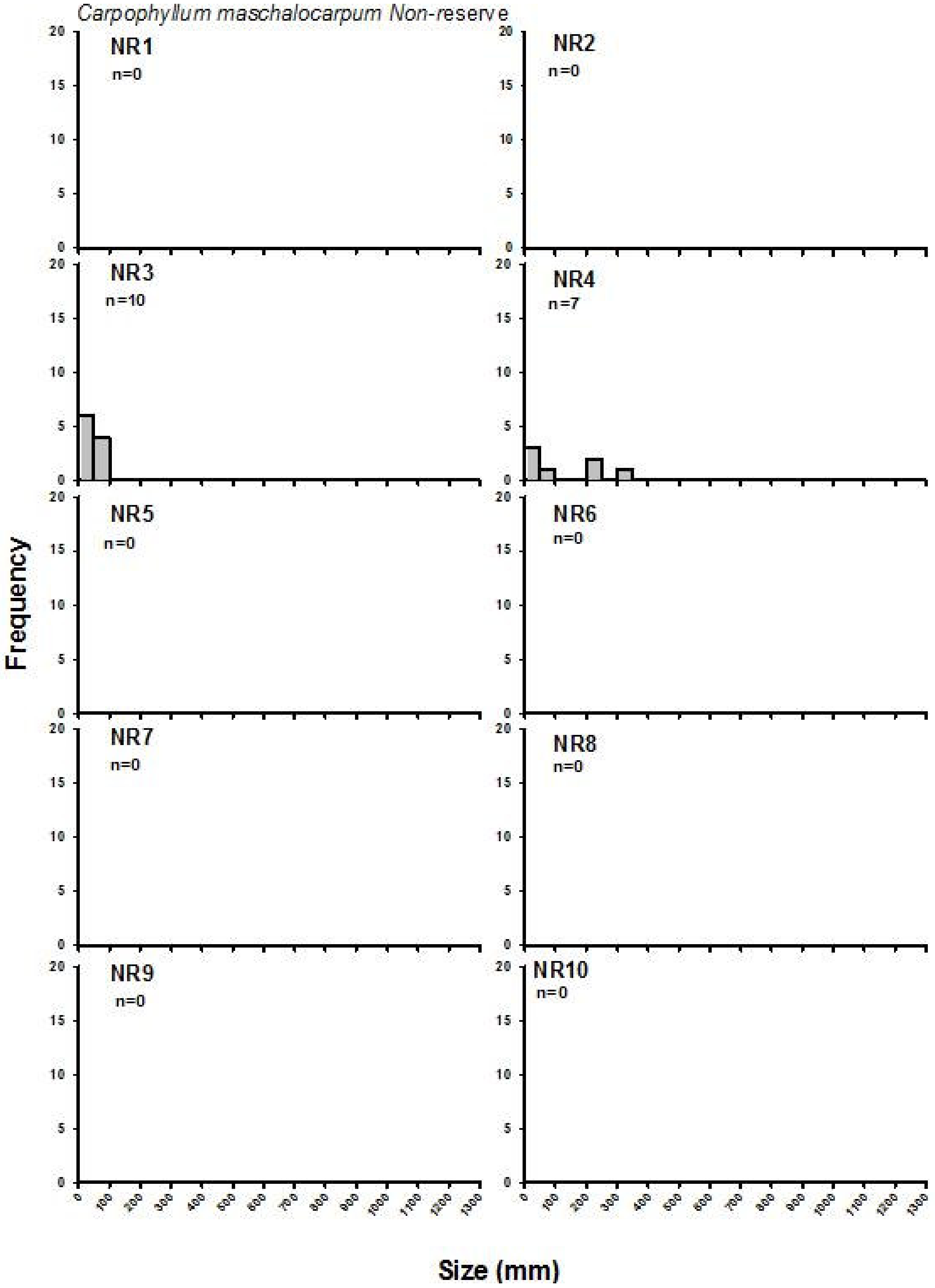

Figure A1 Continued. Size frequency distribution of *Carpophyllum maschalocarpum* within non-reserve sampling sites in 2006.

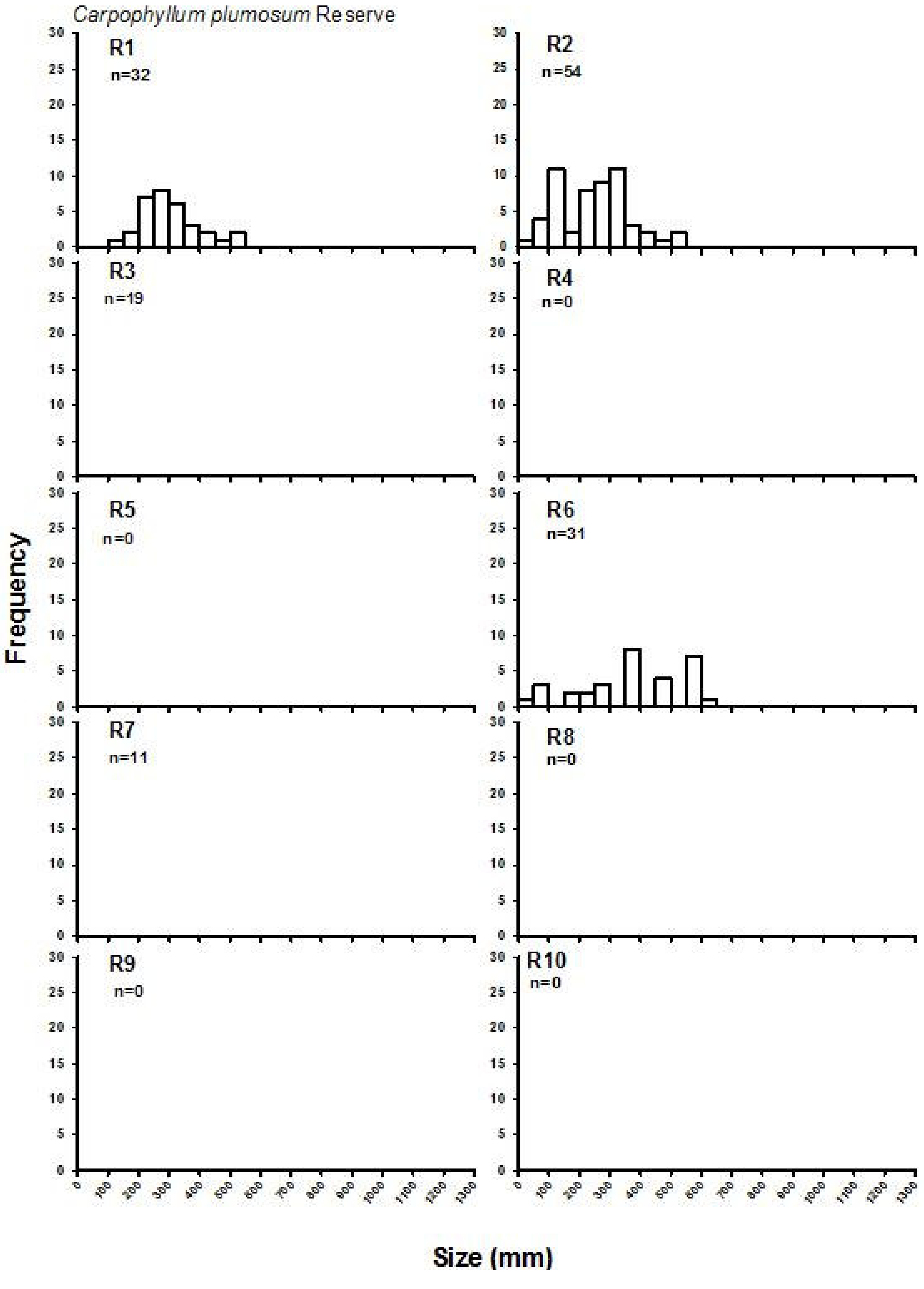

Figure A1 Continued. Size frequency distribution of *Carpophyllum plumosum* within reserve sampling sites in 2006.

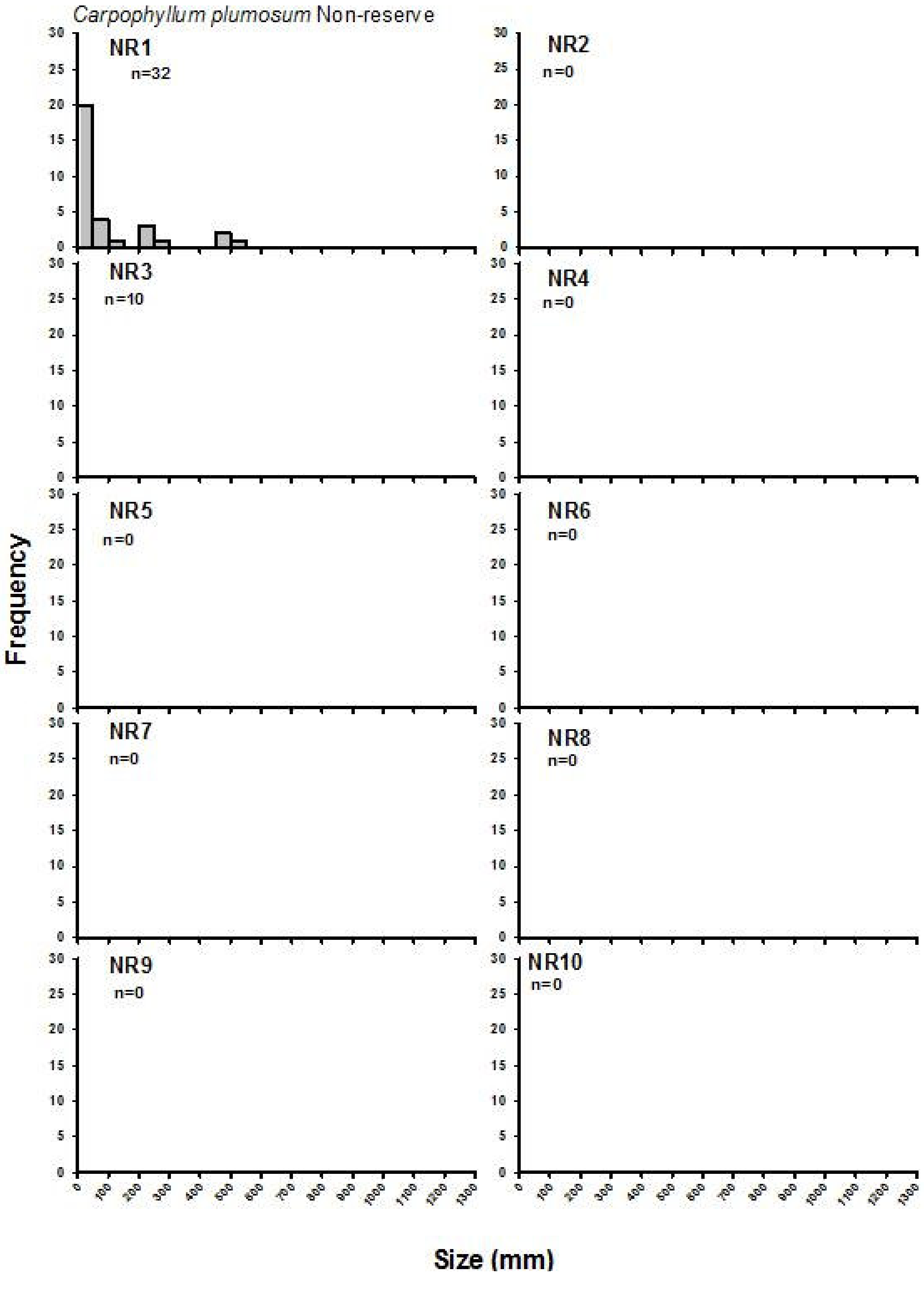

Figure A1 Continued. Size frequency distribution of *Carpophyllum plumosum* within non-reserve sampling sites in 2006.

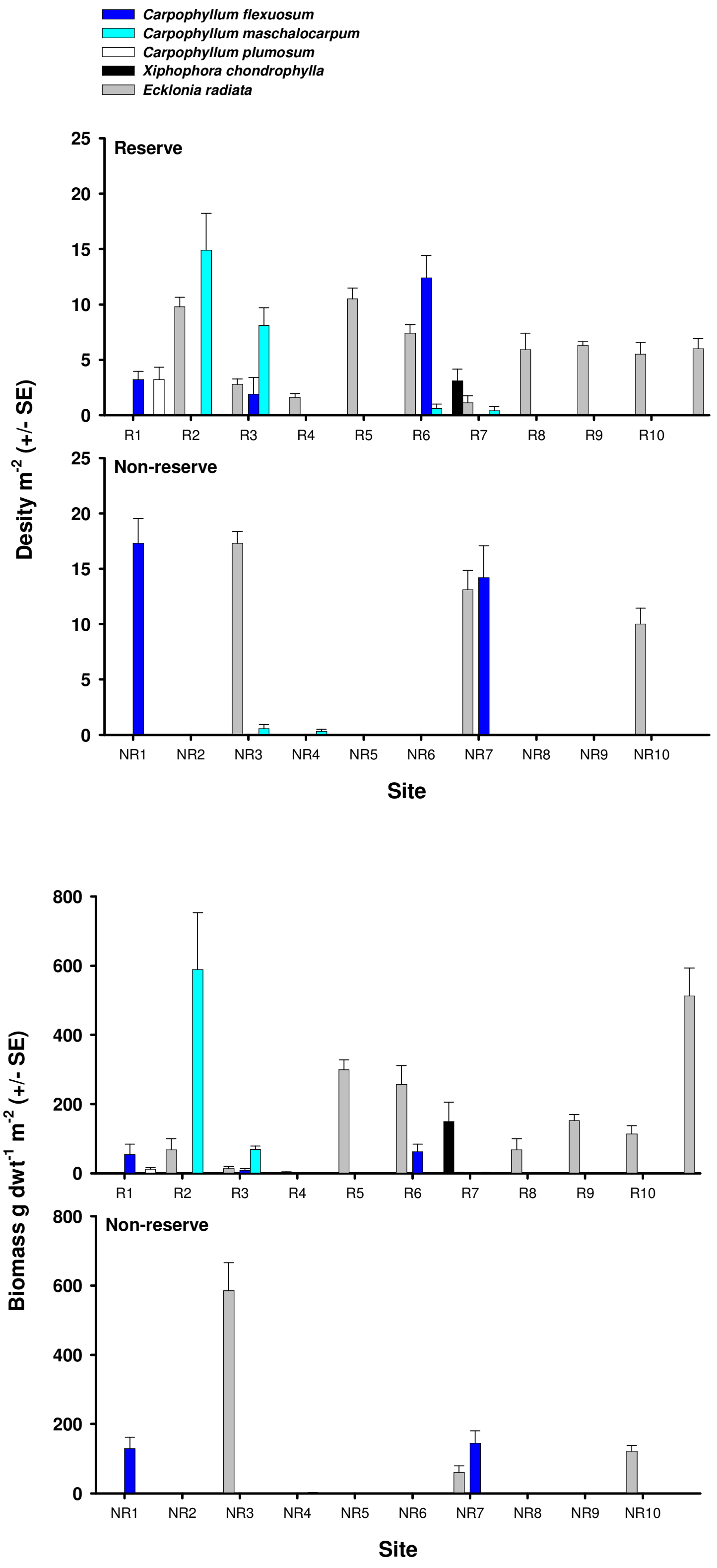

Figure A.3 Mean density and biomass of dominant macroalgae within Te Whanganui-a-Hei Marine Reserve and non-reserve areas sampled. Refer to Fig. 2.1 for sampling sites.

Appendix 2.0

Table A1. Presence (+) and absence (blank) of dominant algal taxa across reserve (R1-R10) and non-reserve (NR1-NR10) sites in 2009.

Taxa	R1	R2	R3	R4	R5	R6	R7	R8	R9	R10	NR1	NR2	NR3	NR4	NR5	NR6	NR7	NR8	NR9	NR10
Ecklonia radiata	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+
Carpophyllum maschalocarpum	+	+	+			+	+				+			+	+	+	+			
Carpophyllum plumosum	+	+	+			+					+		+		+			+		
Carpophyllum flexuosum	+	+	+				+				+		+	+	+	+	+		+	+
Xiphophora chondrophylla	+		+			+					+		+					+		
Landsburgia quercifolia		+																		
Lessonia variegata														+						
Halopteris spp.											+									
Zonaria turneriana	+	+	+	+	+		+	+	+	+		+	+							
Dictyota sp.									+	+										+
Sargassum sinclairii	+																	+		
Carpomitra costata																				
Padina australis				+	+			+	+	+		+								+
Ralfsia sp.		+		+	+	+		+	+	+	+	+		+	+	+	+	+	+	+
Osmundaria colensoi	+		+								+									
Melanthalia abscissa			+	+	+		+	+	+	+		+		+						
Pterocladia lucida	+		+				+				+		+	+	+	+	+		+	
Pterocladia capillacea		+		+	+			+	+	+		+								+
Champia sp.							+			+	+				+	+				+
Curdiea coriacea					+			+	+	+		+							+	+
Coralline turf	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+
Codium convolutum	+	+	+	+	+	+	+		+	+	+	+	+	+	+	+	+	+	+	+
Crustose coralline	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+
Red turf 1	+	+	+	+	+	+		+		+	+	+			+	+	+			+
Red turf 2					+	+			+		+				+			+		

Table A2. Presence (+) and absence (blank) of dominant molluscan taxa across reserve (R1-R10) and non-reserve (NR1-NR10) sites in 2009.

Taxa	R1	R2	R3	R4	R5	R6	R7	R8	R9	R10	NR1	NR2	NR3	NR4	NR5	NR6	NR7	NR8	NR9	NR10
Cookia sulcata	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+
Trochus viridis	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+
Cantharidus purpureus	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+
Modelia granosa			+	+	+		+	+	+	+	+	+	+	+	+		+	+	+	+
Calliostoma punctulatum																				+
Calliostoma tigris	+			+		+	+	+	+		+	+			+	+	+			
Buccinulum linea	+		+	+				+	+	+	+	+	+	+	+		+			+
Astraea heliotropium					+			+	+											
Muricopsis octogonus				+	+			+												
Cellana stellifera			+								+				+			+		
Haliotis iris			+								+			+			+	+		
Haliotis australis											+									
Xymenella sp1 (red foot)		+	+	+		+	+	+	+	+		+	+	+	+	+	+	+		
Xymenella sp2 (orange foot)			+	+	+	+		+	+	+		+	+	+			+	+	+	
Xymenella sp3 (white foot)			+	+		+	+						+							
Cominella quoyana									+	+	+		+		+		+	+		
Cominella virgata	+		+				+			+							+			+
Maoricolpus roseus	+																			
Cabestana spengleri		+					+					+		+		+				
Dicathais orbita																				
Penion sp.					+							+								
Charonia lampas					+															
Ceratosoma amoena																				
Diloma sp.	+																			
Micrelenchus sp.		+				+	+	+				+							+	
Scutus breviculus																				+
Ceratosoma amoena		+		+	+			+	+		+	+		+		+			+	+
Jason mirabilis										+										+

Table A3. Presence (+) and absence (blank) of dominant sessile invertebrate taxa across reserve (R1-R10) and non-reserve (NR1-NR10) sites in 2009.

Taxa	R1	R2	R3	R4	R5	R6	R7	R8	R9	R10	NR1	NR2	NR3	NR4	NR5	NR6	NR7	NR8	NR9	NR10
Tethya aurantium	+	+	+	+				+	+	+	+	+	+	+	+		+		+	+
Tethya ingalli	+	+	+	+	+		+	+	+	+	+					+			+	
Tethya sp. *(3)(compacta?)*	+	+	+	+	+	+	+	+	+	+	+	+	+	+		+	+	+	+	+
Aaptos aaptos								+	+	+										+
Ancorina alata				+	+				+	+		+	+		+				+	
Cliona celata	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+		+	+
Stelletta sp 1					+								+	+	+	+				
Stelletta sp 2				+								+							+	+
Dysidea sp.		+						+	+	+		+								+
Polymastia granulosa		+		+				+	+	+		+								+
Polymastia sp.		+						+	+	+						+		+	+	
Polymastia fusca		+						+	+	+										+
Ciocalypta polymastia		+		+				+	+		+	+							+	+
Microciona sp. (bright red or orange)			+	+			+	+	+	+						+		+		+
Hymedesmia sp (bright orange)	+		+	+				+			+					+			+	+
Raspailia topsenti				+	+			+	+	+		+							+	+
Psammocinia hawere				+	+							+								+
Geodia regina	+			+	+				+	+		+								+
Biemna novaezelandiae				+					+	+			+							+
Chelonaplysilla violacea				+				+			+	+							+	
Tetrapocillon novaezealandiae	+			+				+	+	+								+		+
Callyspongia ramosa									+	+		+								+
Aplysilla rosea								+						+						+
Aplysilla sulphurea				+					+	+			+	+						+
Cnemidocarpa bicornuta	+	+		+			+	+			+	+	+	+	+	+	+	+	+	
Asterocarpa coerulea	+	+	+	+	+	+	+	+	+	+	+	+		+	+	+	+		+	+
Asterocarpa humilis																+				
Aplidium knoxi								+	+	+		+						+		+
Aplidium adamsi	+	+	+	+	+	+	+	+	+	+	+	+		+		+	+		+	+
Aplidium novaezelandiae	+					+	+				+	+	+	+	+		+			
Psuedodistoma novaezelandiae	+	+	+	+		+	+	+	+	+	+	+	+	+	+	+	+	+	+	+
Diademnum densum	+	+		+				+	+	+	+	+			+					
Diademnum novaezelandiae														+						
Diademnum vexillum				+				+	+	+		+								+
Botrylloides sp.					+	+	+		+	+									+	+
Alloecarpa minuta					+	+	+		+	+		+							+	

Solanderia sp.						+	+	+		+		+								+
Sertularia sp.				+	+			+	+		+	+								
Aglaophenia laxa				+				+	+	+		+								+
Aglaophenia acanthocarpa								+		+										+
Alcyonium aurantiacum					+			+	+	+										+
Monomyces rubrum		+		+	+			+	+	+		+	+						+	
Culicia rubeola	+		+	+	+	+	+	+	+	+	+	+		+	+	+			+	+
Actinothoe albocincta		+		+	+	+	+	+	+	+		+			+			+		+
*Corynactis haddon*i	+						+		+	+						+				+
Steginoporella neozelanica				+				+	+	+									+	+
Hornera sp.				+				+	+			+							+	+
Catenicellidae sp.										+										+
Terebratella sp.				+				+	+		+	+							+	

Appendix 3.0

Table A4. Lobster and benthic monitoring co-ordinates.

Site name	Status	Depth	Easting	Northing	Data set
R1	reserve	4.0	2757520	6483072	Historical (5S)
R2	reserve	4.6	2758637	6482757	Historical (2S)
R3	reserve	3.8	2759565	6481925	New in 2006
R4	reserve	11.5	2759425	6483523	Historical (2D)
R5	reserve	17.3	2759535	6483705	New in 2006
R6	reserve	4.6	2760130	6483068	Historical (3S)
R7	reserve	9.7	2760493	6483188	Historical (3D)
R8	reserve	13.2	2761047	6482452	New in 2006
R9	reserve	15.1	2759945	6483125	New in 2006
R10	reserve	17.4	2759270	6483784	New in 2006
NR1	non-reserve	4.6	2757523	6485051	Historical (1S)
NR2	non-reserve	12.8	2757866	6485183	Historical (1D)
NR3	non-reserve	7.4	2761825	6480855	New in 2006
NR4	non-reserve	4.3	2761773	6480440	New in 2006
NR5	non-reserve	6.2	2761797	6480286	Historical (4S)
NR6	non-reserve	10.1	2761921	6480264	New in 2006
NR7	non-reserve	7.4	2761852	6479580	New in 2006
NR8	non-reserve	3.5	2762081	6479222	New in 2006
NR9	non-reserve	11.6	2761830	6481547	Historical (4D)
NR10	non-reserve	14.7	2761618	6483323	New in 2006