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Executive Summary

The Long Bay Marine Monitoring Programme (LBMMP) was initiated in 1998 with the objective of detecting and documenting the impact of urban development on the intertidal and subtidal marine environments of Long Bay. In 1999 the programme was extended to other coastal areas in order to place any patterns observed at Long Bay into a regional context. Subtidal bays monitored include: Waiwera, Stanmore Bay, Little Manly, Torbay and Campbells Bay.

Of particular concern to the Auckland Regional Council (ARC) are the potential impacts of increased sedimentation with changes in land-use through time. Therefore, sediment traps deployed at each of the subtidal sites are monitored to provide an indication of changes in water-borne sediment characteristics as development proceeds. This report presents the findings of monitoring carried out at these locations between September 2005 and March 2008 and describes observed patterns and trends in communities from 1999 to 2008.

The subtidal community structure was strongly correlated by exposure (as measured by wind fetch); exposure explained more spatial variation in the assemblage than the north-south bay-level gradient examined in the previous LBMMP report (Anderson et al. 2005). The communities at Long Bay fall within the exposure gradient and were broadly similar to those found at other bays (Torbay, Stanmore and Campbells) in the inner Hauraki Gulf. After exposure, the most important environmental variables influencing biotic assemblage structure (in order) were: the average rate of trapped fine sediment at each bay, the El Niño Southern Oscillation (ENSO) from the preceding three months and the percentage cover of sediment on the substratum.

The most significant change in community structure since 1999 was a shift in the dominant species of brown alga, from *Carpophyllum maschalocarpum* to *C. flexuosum* (throughout all bays). Notably, however, there was no evidence of any changes in algal biomass. Overall increases were seen in the abundances of the alga *Zonaria turneriana*, the sponge *Tethya burtoni*, and solitary ascidians.

There is currently little or no evidence to suggest that the subtidal Long Bay biota have been impacted in any significant or negative way since the start of the monitoring programme. The subtidal communities have remained relatively stable and minor temporal trends observed there have also been observed at other monitored bays as part of general regional trends.

Importantly, the results indicate that this monitoring programme is sensitive and has the power to detect relevant changes in community structure through time across the entire region, and at particular bays, even within the current pre-development phase. As monitoring continues, we anticipate the ability to more accurately quantify those environmental variables that help determine the abundance and structure of the biota in LBMMP. In particular, continued monitoring will aid with the quantification of depth and climate effects, and should help in determining the relationship between trapped sediment variables and biotic changes.
Introduction

Long Bay is situated immediately to the south of the Whangaparoa Peninsula and north of the residential areas of the North Shore and East Coast Bays (20 minutes’ drive north of Auckland’s CBD). The Long Bay Marine Reserve was established in 1995 and extends 926 metres offshore, stretching from Toroa Point in the south to the Okura Estuary in the north. Long Bay has a diverse array of habitat types, including sandy beaches, intertidal reefs, shallow subtidal reefs, deeper soft-bottom areas and muddy estuarine habitats, making it a representative sub-sample of the habitat types in the greater Hauraki Gulf area (Walker et al. 2001).

The Long Bay catchment has been re-zoned to be within metropolitan urban limits (Auckland Regional Growth Forum 1999). This re-zoning allows for urban development along the cliff tops above the beach and along the streams, in particular Awaruku stream. Monitoring was initiated due to concerns over terrestrial sediment entering the Long Bay marine environment during or after heavy rain events via either Awaruku (at the southern end of the beach) or Vaughans (at the northern end) streams as well as via the Okura and Weiti estuaries. The Long Bay Marine Monitoring Programme (LBMMP) was initiated in 1998 with the objective of detecting and documenting the impact of urban development on the intertidal and subtidal marine environments of Long Bay. In 1999, the programme was extended to other coastal areas around Long Bay. This spread of sites ensures that any impact detected at Long Bay can be placed within the appropriate regional context, given the likely spread of sediment from catchment urbanization along the coast (Green et al. 2000). These other bays would either act as control sites, or could be used to assess the spread of disturbance at various distances away from Long Bay.

Originally, the LBMMP monitored subtidal rocky reef epifauna and flora and beach infauna. However, following a project review, the beach component was discarded due to the low likelihood of sedimentation having an impact in this environment (Haggitt & Mead 2006). A summary of the results of the intertidal beach sampling until 2005 can be seen in Anderson et al. (2005).

2.1 Sedimentation effects upon rocky reefs

Sedimentation has been identified internationally as an increasing and widespread problem for a range of habitats (Airoldi 2003). Human activities on land have greatly accelerated rates of sedimentation and infilling of estuaries (Thrush et al. 2004). In New Zealand, lethal and sub-lethal effects have been documented as a result of sedimentation on soft-sediment macrofaunal communities and much of this research is summarized in a report by Gibbs and Hewitt (2004). In addition, much work has been completed in soft-sediment habitats using present day distributions of organisms along sediment textural gradients to predict the consequences of increased sediment deposition upon community structure (Thrush et al. 2003, Ellis et al. 2006, Anderson et al. 2007).

Some of this increased sediment load will either be discharged directly at the coast, or exit via estuaries, where it may be deposited on subtidal reefs. Increased
sedimentation on subtidal reefs may smother or scour adult and juvenile stages of organisms and prevent settlement of propagules (see review in Airoldi 2003 and references therein). Sedimentation may play an important structuring role in subtidal algal communities on temperate rocky reefs (Schiel & Foster 1986, Schaffer & Parks 1994, Dayton 1995, Renaud et al. 1996, Airoldi & Cinelli 1997, Airoldi & Virgilio 1998, Irving & Connell 2002a, Irving & Connell 2002b, Airoldi 2003, Balata et al. 2007). In general, algal diversity is reduced (Gorostiaga et al. 1998) and there is a loss of canopy-forming seaweeds at sites with high sediment loads (Airoldi 2003). In addition, recent research suggests that settlement and survival of the habitat forming urchin, *Evechinus chloroticus*, can be negatively affected by low levels of sediment deposition (Walker 2007).

2.2 Models and hypotheses

This report will:

1. analyse trends in the biological data from 1999 - 2008 (when biotic methods were comparable), and trends in sediment data from 2002 - 2008 (when sediment processing methods were comparable),

2. determine whether changes in biota are linked to changes in environmental variables,

3. place the monitoring results thus far into an appropriate spatial and temporal context, and

4. use the data collected to predict the consequences of increased sedimentation for subtidal rocky reef communities.
Methods

A chronological synopsis of changes in methods since the inception of the programme is given in the Appendices (section 7.1).

3.1 Site locations

Six bays are sampled in the LBMMP (from north to south): Waiwera (W), Stanmore (S), Little Manly (M), Long Bay (L), Torbay (T) and Campbells Bay (C), and within each bay, five sites are sampled, for a total of thirty sites. The locations of each site are shown in Figure 1 and their GPS coordinates are given in Appendix B (section 7.2). The sites are all situated on rocky subtidal reefs dominated by macroalgae at depths of less than 4.2 m (below chart datum). Sites are located both to the north and south of Long Bay in order to have appropriate spatial controls.

Figure 1
Map showing location of study sites. Bold font indicates study bays (with abbreviations) and italics indicate place names of note. Stars and numbers and names on the water indicate subtidal sampling sites.

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For brevity, bays will be referred to as: Waiwera, Stanmore, Manly, Long Bay, Torbay and Campbells.
3.2 Biotic data

Sampling of subtidal assemblages has been conducted annually in February/March from 1999 onwards at each site (Figure 1). At each site, \( n = 7 \) replicate quadrats, each measuring 1 m\(^2\) were placed haphazardly at each site within 20 m of site markers.

In each quadrat, individual species of macroalgae and macroinvertebrates (> 5 mm) were counted and identified to the lowest practicable taxonomic level (generally to species). Highly mobile organisms such as crabs were not included in counts. In addition, the percentage cover of the substratum made up of different components (such as turfing algae, large brown algae, encrusting coralline algae, bare rock, sediment, sponges, etc.) was estimated visually. Percentage cover was estimated for the smallest components first, using a 10 cm × 10 cm area as 1 % cover, until only one cover type remained. For uneven topography the percent cover measures were scaled accordingly, e.g., if an overhang meant there was an extra area of approximately 50% under the quadrat, then an area of 15 cm × 10 cm was used to estimate 1 % cover. In five of the seven quadrats, the size of each individual was measured, to the nearest 5 cm (using a tape measure) for macroalgae and to the nearest millimetre (using a ruled bar) for macroinvertebrates. In the other 2 of the 7 quadrats macroalgae and macroinvertebrates were counted but not measured. Although sampling at Long Bay originated in 1998 (Babcock et al. 1998), methodology has generally been consistent only since 1999. Details of changes to sampling methodology through time are documented in the Appendices (section 7.1).

3.3 Environmental data

3.3.1 Fetch

Fetch was calculated for each central bay position by averaging distance to land for each 10-degree sector of the compass rose (see Fletcher et al. 2005). Following Shears and Babcock (2007), the radial maximum distance was arbitrarily set to be 300 km for open sectors of water. Bays ranked by fetch (most to least exposed) were:

1. Waiwera (average fetch = 15.86 km)
2. Campbells (average fetch = 14.77 km)
3. Stanmore (average fetch = 13.97 km)
4. Torbay (average fetch = 13.61 km)
5. Long Bay (average fetch = 12.86 km)
6. Manly (average fetch = 9.06 km).
3.3.2 Sediment

An important goal of the LBMMP is to attempt to relate any potential changes in assemblages over time to changes in sedimentation regime. Direct measures of sedimentation are extremely difficult and costly to obtain, due to the natural movement of marine and terrestrially-derived sediments, resuspension with currents and wave action. A proxy measure of sediment fluxes is provided by sediment traps. Sediment traps were deployed at each of the 30 sites (5 sites × 6 bays) which were designed to retain sediment deposited from the water column and to avoid potential resuspension (see Walker et al. 2001 and Ford et al. 2003a for details).

Sediment in traps were collected approximately monthly, however, there were often problems retrieving samples due to bad weather or loss of traps through public interference. Thus, the number of each site’s trap samples is highly variable per year (from \( n = 0 - 11 \)); as is the length of time that traps were left between sampling times. After collection, the contents of traps were filtered through a 1.2 \( \mu \text{m} \) filter, oven dried at 65 - 80°C for 24 hours (or until dry), then weighed. Values for the rate of sediment accumulation in each trap at each successful time of sampling were calculated in grams per cm\(^2\) per day. Some traps were filled to the extent that sediment retention within traps could not be guaranteed (aspect ratio < 7), therefore high values for trapped sediment should be regarded as conservative, i.e., less than was likely to have occurred.

Sediment traps provide a measure of sediment deposition or flux at a site, but the high-aspect ratio means that it is likely to preclude within-trap resuspension. However, traps may also capture sediment suspended locally from the neighbouring bed, so sediment deposition in traps is referred to as ‘trap rate’ within this document.

The grain size characteristics of sediments in traps are also of interest. It is expected that increases in sedimentation due to changes in land-use would result in increases in the rate of fine sediments deposited (i.e., mud less than 63 \( \mu \text{m} \) in diameter). The proportion of the volume of sediments obtained in each trap that is less than 63 \( \mu \text{m} \) in diameter has been measured since mid-2002 using a Galai laser particle analyser. Any resuspension from within traps is likely to be grain size specific; therefore traps with aspect ratios less than 5 before June 2002 and less than 7 after June 2002 were excluded from textural analyses. This difference in aspect ratio is driven by the difference in trap design prior to June 2002 (Appendix A - section 7.1) which means that greater confidence can be had in textural recordings from after that time. From 2003 onwards, each sample was also pre-treated to remove organics according to the recommendations and methods provided in Ford et al. (2003b). Note that data from 2001 onwards can be used for statistical analyses without adjustment, as the relationship between values obtained with versus without pre-treatment is almost 1:1 (further details in Ford et al. 2003b). The possible artefact inherent in this lack of adjustment will be discussed in light of overall trends in the results.

Trapped sediment variables used in analyses (see section 3.4) were:

(i) the average trapped sediment rate (measured in grams per cm\(^2\) per day).
(ii) the proportion of trapped fine sediments (less than 63 \( \mu \text{m} \)).
(iii) the rate of deposition of fines in traps (measured in grams per cm\(^2\) per day).
When the trapped sediment variables were applied as an explanatory covariate (to estimate their effect on biotic abundance or assemblage structure), they were averaged at the site level across each year prior to biotic sampling. When trapped sediment variables were examined and tested for changes over time, they were averaged at the bay level for each month.

3.3.3 Depth

Many of the key species in the LBMM are known to have relatively narrow distribution zones within sites. For example, macroalgae within the inner Hauraki Gulf have distinct but narrow distributions - *Carpophyllum maschalocarpum* often forms a narrow band < 3 m depth, mixed stands of algae are dominant between 2 - 5 m depth (Walker 1999, Shears 2003), and stands of *Ecklonia radiata* and *Carpophyllum flexuosum* are abundant > 5 m depth. These types of species gradients are due to the interplay of physical variables (e.g. turbidity, exposure) (Grace 1983; Walker 1999).

Haggitt and Mead (2006) were concerned that patterns in *C. flexuosum*, *C. maschalocarpum*, *Cantharidus purpureus*, *Dicathais orbita*, solitary ascidians and *Tethya burtoni* seen at sites in Stanmore and Waiwera may, in fact, be due to depth changes over time, rather than real biological change at a constant depth. To address these issues, tidally corrected depth was included as a potential covariate in both univariate and multivariate models. Models used depth records to quantify and remove the depth distribution effects from the overall analyses (see section 3.4.2.2); all univariate model results quantified the changes in abundance having accounted for changes in depth over time.

**Depth records**

Depth for each subtidal quadrat was recorded during faunal sampling between 1999 - 2002, 2007 and 2008. Sporadic quadrat depth records also exist for 2005 (in Torbay and Long Bay). Site depth records for the 2006 survey used the trap survey depths recorded in November 2005 (several months prior to the 2006 faunal survey). Although these measurements did not contain quadrat scale measurements of depth, they did contain the average estimate of depth for each site. This site average was used as a proxy for quadrat depth for missing values in the 2006 survey.

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2 For simplicity, every taxon will generally be referred to by its genus after an initial introduction. The exceptions to this are the *Carpophyllum* fucales (referred to as *C. maschalocarpum*, *C. flexuosum* and *C. plumosum*) and the *Cominella* gastropods (*C. adspersa*, *C. glandiformis*, *C. virgata* and *C. maculosi*).

3 Trap depth has changed over time when traps were lost (after being lost due to anthropogenic/climactic interference) or dragged and replaced in slightly different positions.
Table 1
Recorded average site depths (meters above chart datum) for each bay in the LBMMMP. Depth values was rarely recorded between 2003 and 2005 (exceptions being Torbay and Long Bay in 2005).

<table>
<thead>
<tr>
<th>Year</th>
<th>Campbells</th>
<th>Torbay</th>
<th>Long Bay</th>
<th>Manly</th>
<th>Stanmore</th>
<th>Waiwera</th>
</tr>
</thead>
<tbody>
<tr>
<td>1999</td>
<td>0.3</td>
<td>0.2</td>
<td>0.0</td>
<td>0.3</td>
<td>-0.2</td>
<td>-0.1</td>
</tr>
<tr>
<td>2000</td>
<td>1</td>
<td>0.1</td>
<td>0.3</td>
<td>-0.2</td>
<td>0.8</td>
<td>-0.1</td>
</tr>
<tr>
<td>2001</td>
<td>-0.3</td>
<td>0.6</td>
<td>-0.1</td>
<td>-0.1</td>
<td>-0.3</td>
<td>0.5</td>
</tr>
<tr>
<td>2002</td>
<td>0.2</td>
<td>-0.3</td>
<td>-0.4</td>
<td>0.0</td>
<td>1.0</td>
<td>0.5</td>
</tr>
<tr>
<td>2003</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>2004</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>2005</td>
<td></td>
<td></td>
<td>0.3</td>
<td>0.4</td>
<td></td>
<td></td>
</tr>
<tr>
<td>2006</td>
<td>0.5*</td>
<td>1.3*</td>
<td>1.0*</td>
<td>1.4*</td>
<td>1.3*</td>
<td>1.5*</td>
</tr>
<tr>
<td>2007</td>
<td>1.1</td>
<td>0.2</td>
<td>0.8</td>
<td>0.7</td>
<td>1.2</td>
<td>1.5</td>
</tr>
<tr>
<td>2008</td>
<td>1.4</td>
<td>1.0</td>
<td>1.5</td>
<td>1.1</td>
<td>1.4</td>
<td>1.3</td>
</tr>
</tbody>
</table>

*Indicates the average pre-survey site depths taken in November 2005 (average site values are used for each quadrat for the 2005 data).

3.3.4 Climate - the El Niño Southern Oscillation (ENSO)

The “El Niño Southern Oscillation” (ENSO) phenomenon is the most important coupled ocean-atmosphere phenomenon to cause global climate variability on an inter-annual time scale (Vasquez et al. 2006). ENSO refers to the major systematic global climate fluctuation that occurs at the time of an “ocean warming” event, and ‘El Niño’ and ‘La Niña’ events refer to opposite extremes of the ENSO cycle. It is known to have an important influence on New Zealand’s climate (Gordon 1986) and may also be a useful predictor of organism abundances.

During El Niño events, New Zealand tends to experience stronger or more frequent winds from the west in summer, typically leading to drought in East coast areas and more rain in the West. In winter, the winds tend to be more from the south, bringing colder conditions to both the land and the surrounding ocean. In spring and autumn south-westerly winds are more common. In contrast, La Niña events are characterized by north-easterly winds which tend to bring moist, rainy conditions to the North-East of the North Island, and reduced rainfall to the South and South-West of the South Island. Warmer than normal temperatures typically occur over much of the country during La Niña, although there are regional and seasonal exceptions (Gordon 1986, Mullen 1996).

To model ENSO, we chose a Multivariate ENSO Index (MEI) that is derived from tropical Pacific COADS (Comprehensive Ocean Atmospheres dataset) records. The MEI is derived from the first principal component of six variables: sea level pressure, surface zonal wind components, meridional wind components, sea surface and surface air temperature, and cloudiness. The index data is available at: [http://www.cdc.noaa.gov/people/klaus.wolter/MEI/table.html](http://www.cdc.noaa.gov/people/klaus.wolter/MEI/table.html).
We felt that ENSO effects might vary depending on the temporal scale of the ENSO signal, so MEI values were averaged over various different time scales. A multivariate analysis (a distance based linear model (see section 3.5.1)), showed that the average of the preceding 3 months ENSO values explained the greatest variation in the biotic assemblage over time; so univariate models used this period for the ENSO term. In short, ENSO was modelled using the MEI on two different time scales:

1. The month prior to trap collection (when correlating ENSO changes with sub-tidal trap data changes).
2. The 3 month period preceding biotic sampling (used to correlate with biotic data).^4

### 3.4 Statistical Methods

All multivariate analyses of community structure were done using the PRIMER v6 computer program (Clarke and Gorley 2006) with the PERMANOVA+ add-on package (Anderson and Gorley 2008). In addition, all multivariate analyses were done on the basis of Bray-Curtis dissimilarities calculated from log(x+1) transformed abundances. All univariate analyses and plots were done using the R package for statistical computing (R Core Development Team 2007).

#### 3.4.1 Characterising the bays

‘Similarity percentages’ were calculated across all bays in the LBMMP (for each year’s data) for both count and cover data. This was implemented using the SIMPER routine, which decomposed average Bray-Curtis similarities between year by bay combinations into percentage contributions from each species. To visualize the results, the Bray-Curtis similarities were plotted as dendrograms.

Due to changes in assemblages over time, classification trees were used to find those variables (or levels of particular variables) that discriminated between bays in the LBMMP (Breiman et al. 1984, Ripley 1996). Classification trees are a data mining technique used to predict membership of cases (quadrats) in the classes of a categorical dependent variable (bays), from their measurements of one or more predictor variables (e.g. biotic and environmental variables). They systematically try to split samples into two parts, examining one predictor variable at a time and splitting the records on the basis of a dividing line in that variable (e.g., *Trochus viridus* abundance > 11 or ‘sediment cover’ <= 35%).

To use the classification trees, a separate binary factor was created for each bay. This factor identified ‘bay membership’ as: (i) the bay of choice (e.g., ‘Waiwera’) vs (ii) ‘other’, (e.g. all other bays that were not Waiwera). Each classification tree used this factor as a response, thereby attempting to find those predictor variables that differed between the bay of choice and all other bays.

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^4 It should be noted that the month of subtidal sampling differed between years due to restrictive weather conditions.
3.4.2 Assessment of changes over time

3.4.2.1 Multivariate models assessing change through time

Change over time can be characterized either by magnitude (dispersion) or direction (seriation). For example, a bay may be highly variable over time, but in no particular direction; alternatively a bay may change relatively little, but in a consistent direction. To test these facets of temporal change, we examined both multivariate dispersion (variability over time) and seriation (directional change over time).

Visualizing patterns

To visualize patterns, non-metric multi-dimensional scaling (MDS) ordination was done on centroids (averages) for year by bay combinations. Each MDS plot showed associated biplots, showing vectors that corresponded to strong correlations (\( \rho < 0.4 \) absolute) of individual taxa to axes on the plot. The Bray-Curtis dissimilarities among main effects centroids [i.e. 6 bays (averaged across all times) + 10 time points (averaged across all bays)] were calculated after \( \ln \) transformation and plotted using MDS. Vectors that corresponded to strong correlations (\( \rho < 0.75 \) absolute) between individual taxa and the PCO axis were also visualized using biplots.

Temporal dispersion

To measure the temporal variation in community structure among bays, the abundances of individual taxa were averaged across the \( n = 7 \) quadrats for each site at each time point. Patterns of dissimilarity among the bays and through time were visualized using non-metric multi-dimensional scaling (MDS) of the bay x time centroids. The temporal variability (dispersion) of each bay was quantified as the average Bray-Curtis dissimilarity among time points. These dispersions were formally compared among the bays using a permutation test of dispersion with 4999 permutations (PERMDISP, Anderson 2006).

Temporal seriation

The hypothesis of a gradient through time (a multivariate pattern of seriation in years) was investigated for each bay (count and cover data) using a Mantel’s test (Mantel 1967) using Spearman’s rank correlation (\( \rho \)) available in the RELATE routine of PRIMER (Clarke and Gorley 2006).

The hypothesis of a gradient in biotic assemblage with respect to fetch (i.e. a multivariate pattern of seriation from low to high fetch sites) was also investigated for each year using this routine. All analyses were done separately for counts and percentage cover data. A constrained ordination (CAP) was used to examine the change in assemblage with respect to changes in fetch amongst bays. The CAP analysis used a Bray-Curtis dissimilarity matrix on log(\( x+1 \)) transformed count data.
Size frequency distributions

Temporal impacts can manifest as either changes in abundance, changes in size structure of a population over time, or as a combination of both. Previous analyses have addressed changes in abundance over time; this section will address changes in population size structure over time. The size frequency distributions were examined for the eight most abundant taxa which constituted over 69% of the individuals. Some common species were not examined for changes in population size structure. The brown alga, *Zonaria turneriana*, nearly always fell into the first three size categories (i.e. 5, 10 and 15 cm long), so size frequency analysis was deemed unnecessary. Solitary ascidians and *Tethya* were also common, but were not measured and consequently could not be incorporated in any size frequency analysis.

To guard against spurious correlations, no size frequency data were analysed or plotted for any year by bay combination when the number of size categories exceeded the number of observations.

Size frequency data was calculated proportionately for each bay by year combination, e.g., Waiwera 1999, Stanmore 1999 etc., and then entered into PRIMER with variables being the different size classes e.g. 5, 6, 7 mm etc. However, with *Ecklonia*, stipe length was used as opposed to total length (because total length can be affected by processes such as abrasion or herbivory). The gastropods were deemed to be measured at an insufficient resolution (5 mm intervals) to detect fine-scale ecological change in the review of Haggitt and Mead (2006). Therefore, since 2007 gastropods have been measured at millimetre intervals. In this report gastropod size frequencies will be examined using 5 mm interval data over all ten years of the LBMMMP, and will only be able to detect relatively large scale changes.\(^5\) The relative importance of (i) bay and (ii) year of sampling, on the size frequency distribution of each major measured taxon was examined and quantified by partitioning the total variation in the size frequency distribution using permutational multivariate analysis of variance (PERMANOVA, Anderson 2001; McArdle and Anderson 2001). However, the interaction between bay and year was unable to be tested as only one replicate existed at each combination of these factors.

When a significant change over time was detected, a cluster dendrograms was examined (using PRIMER’s SIMPROF routine). Particular ecological importance was placed upon the presence of juveniles, as juveniles are considered more vulnerable than adults, so their absence could be a precursor of community change.

3.4.2.2 The univariate model assessing change through time

In terms of spatial variation, quadrats within sites were haphazardly allocated and are therefore considered the lowest level of replication and provide estimates of error variation. Summary plots of each site mean tracked through time were produced for every bay. To gain insight into the structure of changes over time, a least squares regression line and a local regression spline smoother were also fit to each bay-level panel.

---

\(^5\) When a longer time series of data has accumulated it is envisaged that data will be examined preferentially using the 1mm interval data.
The change over time (as measured by count abundance and percentage cover) was examined and quantified using Generalized Linear Mixed Models (see the following section). Plots and models were done for the following:

1. The numerically dominant taxa and percentage cover classes.
2. Those taxa that were noted as changing in the previous LBMMP report. This included: the sea cucumber *Australostichopus mollis*, sea star *Patiriella regularis*, the sponge *Tethya burtoni* and herbivorous gastropods *Cookia sulcata*, *Maoricolpus roseus* and *Cantharidus purpureus* (Anderson et al. 2005). Although not numerically dominant, the temporal pattern of *Evechinus* abundance was also visualized, because of its known influence on macroalgal community structure (Andrew 1988).
3. Univariate summary indices, including: the average total number of individuals (per quadrat), the total number of taxa per site (diversity), algal biomass\(^6\), site evenness\(^7\).

*Using a Generalized Linear Mixed Model*

Counts of abundances of organisms are appropriately modelled using a generalized linear model, since:

1. A population of organisms tends to change by multiplying (rather than individuals being added or subtracted). Therefore, rather than fitting a linear (additive) model directly, it is generally more appropriate to model the data on the log (multiplicative) scale.
2. Organisms occur in discrete counts, rather than being a continuous distribution.
3. Abundance data tends to be clumped (aggregated) rather than randomly dispersed.

To account for these factors, each univariate variable was analysed using a quasi-poisson error distribution with log link. Since primary interest was in examining changes across each bay as a whole (rather than at the site level) a Generalized Linear Mixed Model (GLMM, Booth et al. 2003) using the R computer program (R Development Core Team 2005) was used. Specifically, Penalized-Quasi-likelihood was used to estimate model parameters using the ‘*glmmPQL*’ package in the ‘MASS’ library.

The mixed model regarded the five subtidal sites as samples from the general bay area. Note that measures of site diversity, site evenness and percentage cover variables did not demonstrate over-dispersion or a mean-variance relationship and were therefore modelled using a linear mixed-model with normal (Gaussian) errors. Average diversity and evenness values were obtained at the site level, so there were no random effects for these variables.

\(^6\) Algal measurements were converted to biomass using equations described in Shears and Babcock (2003).
\(^7\) Evenness was assessed using “Evar” as recommended by Smith and Bastow Wilson (1996) and calculated using the ecological evenness calculator at: [http://www.nateko.lu.se/personal/benjamin.smith/software/frames.html](http://www.nateko.lu.se/personal/benjamin.smith/software/frames.html)
Of particular interest in the univariate models, was to see if any of the response variables included a significant component of the variable ‘year’, which would indicate a consistent increase or decrease in abundance (or percentage cover) through time. Year effects were estimated after controlling for (‘partialling’) the effects of ENSO (cyclicity) and depth, i.e. the parameter estimating change over time (‘year’) when holding ENSO and depth to be constant.

**Estimating depth effects**

Every bay had a maximum of: 7 quadrats x 5 sites = 35 different depths recorded in each year. The distribution of depths at each bay over time meant that the effect of depth (i.e. the estimated change in abundance as depth increases) could be separated from changes in abundance over time.

**Exploratory Data Analysis**

Exploratory data analysis (EDA) was performed checking the assumptions of the model (i.e. linearity of the response, autocorrelation and heteroscedasticity). Following Bence (1995), if significant temporal autocorrelation was found, the shortness of the time series meant it was modelled by fitting $AR1$ (‘autoregressive errors order one’) errors.

### 3.5 Examining the relationship between temporal change and environmental factors

#### 3.5.1 Multivariate analyses

The relationship between the environmental variables (discussed in section 3.3) and the biotic count data was examined. Unless specified otherwise, all environmental variables were averaged at the bay level for each year prior to biological sampling from 1999 – 2008. Environmental variables examined were:

1. fetch, which is a measure of exposure; variable name: ‘fetch bay’.
2. a measure of ENSO, averaged over the short-term (three months) and medium term (twelve months); variable names: ‘ENSO_st’ and ‘ENSO_med’ respectively.
3. percentage sediment cover; variable name: ‘sed.cover’.
4. trapped sediment variables (see section 3.3.2 for details):
   a. average trap deposition rate (g.cm$^{-2}$.day$^{-1}$); variable name: ‘bay trap_rate’.
   b. percentage fines (<63 µm) in traps; variable name: ‘bay %fines’.*
   c. the rate of deposition of fines in traps; variable name: ‘fines_rate’.*
5. depth (see section 3.3.3 for details).

*The time series for variables using fine sediment is restricted to between 2002 - 2008. This is the period where methodology for measuring fine sediment has been consistent.*

---

*Note: fetch is invariant over time*
These variables were initially examined for approximate normality and possible outliers using draftsman plots. The relationship between the environmental variables and the biotic data was then examined using distance based linear models (DISTLM, Legendre and Anderson 1999, McArdle and Anderson 2001) on the Bray-Curtis dissimilarity matrices of log(x+1) transformed count data.

The initial model was fit using ‘forward selection’ on the $R^2$ criterion to determine the amount of biotic variability that could be explained. The final model was fit by selecting the best out of all possible models using Akaike’s Information Criterion with second order correction for small sample sizes (AICc Akaike 1973, Burnham & Anderson, 2004). To visualize the resulting model, distance-based redundancy analysis (dbRDA, McArdle and Anderson 2001) was used, where the ordination axes are linear combinations of the environmental variables that maximally explain biotic variation.

To estimate the effect of depth on the biotic assemblage, a second DISTLM and dbRDA was done using an expanded environmental covariate dataset. This dataset contained the same variables mentioned above, but additionally had a variable measuring each bay’s averaged tidally corrected depth in each year since 2002 (i.e. the dataset used those interpolated depths mentioned in section 3.3.3). The results were then compared and contrasted against the DISTLM and dbRDA that excluded depth.

### 3.5.2 Univariate analyses

#### Modelling Sediment Trap Rate

Although ENSO and depth variables were included in initial mixed model analyses, respective trapped sediment variables (see section 3.3.2) were not. The rationale behind the initial exclusion of sediment variables was to determine whether abundance was changing over time after removing the effects of ‘natural environmental variables’. When the ‘sedimentation variables’ are included as a covariate, temporal trends in abundance are quantified given the changes in sediment that may also have occurred over time, i.e. the linear trend in abundance may be different because the model might be partitioning some of the changes in abundance to be caused by the effect of sediment. Trapped sediment variables were therefore added sequentially, after: (i) climate, (ii) change over time and (iii) depth effects had been quantified.

When the trapped sediment variables were applied as an explanatory covariate (to estimate their effect on biotic abundance or assemblage structure), they were averaged at the site level across each year prior to biotic sampling. When trapped sediment variables were examined and tested for changes over time, they were averaged at the bay level for each month.

#### Modelling Percentage Sediment Cover

To examine possible changes in component biota with chronic longer-term increases in sediment cover, the most abundant taxa in the subtidal datasets were modelled individually along the percentage sediment cover gradient. Following Anderson et al. (2007) and Ford and Pawley (2008), quantile regression splines of the 95th percentile of the distribution modelled the abundance of dominant species along the regional
gradient of percentage sediment cover using the ‘quantreg’ package in R (Koenker 2007). The appropriate polynomial required for the spline to provide an adequately-shaped fit for each species, was determined using Schwarz’s “Bayesian Information Criterion” (BIC, Schwarz 1978). The value at which the predicted density achieved a maximum along the gradient was identified in order to find the ‘optimal’ percentage cover of sediment for each taxon.
4 Results

4.1 Characterization of the bays

Surveys of the LBMMP from 1999 - 2008 have included 2100 quadrats, encompassing over 206,000 individuals from 69 taxa and 48 different cover types (Appendix D).

The subtidal rocky reef communities between bays are, in general, very similar – dendrograms comparing bays over time shows that even the most different bays have a Bray-Curtis similarity coefficient greater than 65% in both cover and count variables (Appendix E).

4.1.1 The typical LBMMP reef assemblage

A decomposition of the average Bray-Curtis dissimilarity between all pairs of sites in each year was examined using the SIMPER routine within PRIMER. Both count and cover assemblages were relatively similar, with average respective similarities of 58% and 64% (calculated between all sites over all times in the LBMMP). Table 2 shows those species that are typical of sites in the LBMMP. Thus, sites typically have a canopy of brown algae composed of *Carpophyllum maschalocarpum* or *Ecklonia radiata*, which together constitute the majority (more than 65%) of all recorded algae.

The rock below the algal canopy is typically covered in a mixture of crustose coralline algae (over 50% of all the measured cover), sediment (~17%) and the under-story brown alga *Zonaria turneriana* (~3%). A number of macroinvertebrates (mainly grazing echinoderms and gastropods) live on the LBMMP reefs. Of these invertebrates, the gastropod *Turbo smaragdus* (average density of ~18 per quadrat) is approximately five times more numerous than any other taxon.

4.1.1.1 Differences between bays

To find which biota discriminated between bays, six separate classification trees were created. Each of the six trees contrasts a single bay with all the other LBMMP bays aggregated together (‘Other’ – see Appendix F).

Stanmore was found to be comparatively indistinguishable from all other bays in the LBMMP – it required an extremely complex tree to determine bay membership (*cp* =0.004) and had the highest cross-validation error rate, i.e. the expected error rate when predicting bay membership with new data. In contrast, Waiwera and Manly are quite distinct – they have the least complex trees (*cp* = 0.02), and the lowest error rate. Campbells was slightly less distinct than Waiwera or Manly; it had a more complex tree (*cp*=0.015) and higher error rate. Long Bay and Torbay were very similar to each other, and ranked between Campbells and Stanmore in their distinctness.

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9 Note: If opposing value of the same criterion (e.g. a particular biotic abundance level) appear at different levels of a branch, then interactions are likely taking place between the criteria.

10 *cp* = complexity parameter; a measure of tree complexity where lower *cp* values refer to more complex trees.
Based on the classification trees (shown in Appendix F), a ranking of bay distinctness, with a summary of the criteria that distinguish each bay, is shown below:

1. **Waipera**: high (relative) abundance of *Tethya* and *Cantharidus*, but low abundance of *C. maschalocarpum*.

2. **Manly**: low abundance of *Cystophora sp.* and *Zonaria*, but high abundance of *C. plumosum*.

3. **Campbells**: low abundance of *Cystophora, C. plumosum* and *Ecklonia*, but high abundance of *C. virgata* and *C. maschalocarpum*.

4. **Torbay**: low cover of sediment and shell, and low abundance of *Turbo* and *C. virgata*, but high abundance of solitary ascidians and *C. plumosum*.

5. **Long Bay**: Low abundance of gastropods, *C. flexuosum, C. maschalocarpum*, but high sediment cover and abundance of solitary ascidians.

6. **Stanmore**: No simple distinguishing characteristics.

### Table 2
SIMPER results showing those biota (and cover variables) that typify the bays sampled in the LBMMP. Average abundance is shown on the log scale; average similarity shows the average Bray-Curtis similarity between all pairs of sites in the LBMMP. δ/SD is an indication of consistency, i.e. large values (say >1) indicate that the taxa has a consistently large presence. Contrib% shows the percentage of similarity an individual species makes up of the average similarity. Cum% is the cumulative species similarity.

#### Average similarity: 58.4%

<table>
<thead>
<tr>
<th>Count data - species</th>
<th>Average abundance</th>
<th>Average similarity</th>
<th>δ/SD</th>
<th>Contrib%</th>
<th>Cum.%</th>
</tr>
</thead>
<tbody>
<tr>
<td><em>Turbo smaragdus</em></td>
<td>2.39</td>
<td>11.68</td>
<td>2.33</td>
<td>20.00</td>
<td>20.00</td>
</tr>
<tr>
<td><em>Carpophyllum maschalocarpum</em></td>
<td>2.37</td>
<td>10.58</td>
<td>1.44</td>
<td>18.12</td>
<td>38.12</td>
</tr>
<tr>
<td><em>Zonaria turneriana</em></td>
<td>2.30</td>
<td>10.44</td>
<td>1.96</td>
<td>17.88</td>
<td>56.01</td>
</tr>
<tr>
<td><em>Ecklonia radiata</em></td>
<td>1.39</td>
<td>6.32</td>
<td>2.06</td>
<td>10.83</td>
<td>66.83</td>
</tr>
<tr>
<td><em>Trochus viridus</em></td>
<td>1.03</td>
<td>3.87</td>
<td>1.26</td>
<td>6.64</td>
<td>73.47</td>
</tr>
<tr>
<td><em>Carpophyllum flexuosum</em></td>
<td>1.02</td>
<td>2.66</td>
<td>0.63</td>
<td>4.56</td>
<td>78.03</td>
</tr>
</tbody>
</table>

#### Average similarity 64.2%

<table>
<thead>
<tr>
<th>Cover data - variables</th>
<th>Average abundance</th>
<th>Average similarity</th>
<th>δ/SD</th>
<th>Contrib%</th>
<th>Cum.%</th>
</tr>
</thead>
<tbody>
<tr>
<td>Crustose coralline algae</td>
<td>52.86</td>
<td>44.26</td>
<td>3.24</td>
<td>68.91</td>
<td>68.91</td>
</tr>
<tr>
<td>Sediment</td>
<td>16.90</td>
<td>9.54</td>
<td>1.22</td>
<td>14.85</td>
<td>83.76</td>
</tr>
</tbody>
</table>

#### 4.1.12 A gradient in assemblage correlated with exposure

Analyses using PERMANOVA found a gradient between exposure (as measured by wind fetch) and subtidal community structure (); fetch explained more spatial variation in the assemblage than the north-south bay-level gradient examined in the previous LBMMP report (Anderson et al. 2005). The squared canonical correlations (δ²) was very
high, with $\delta^2 = 0.80$ for fetch, indicating a strong correlation between the biotic count data structure and the fetch gradient. The separation of assemblages based on the fetch gradient corroborates the results seen using the classification tree. That is, Waiwera and Manly were relatively distinct, while other bays showed considerable overlap (Figure 2).

A significant fetch by year interaction was found for log($x+1$) transformed abundance, but not for presence/absence data; suggesting that the strength of the fetch gradient varied through time with respect to community abundances, but not community composition (Table 3). Separate individual analyses of the count data showed that the strength of the fetch gradient generally increased until 2007; although the 2008 survey showed a decline (Table 4). In contrast, the overall difference in biotic structure between bays (as measured by the ANOSIM $R$-statistic) erratically increased until 2004, but decreased since that year (Table 4).

A constrained ordination, where the assemblages were ordered along the fetch gradient, elucidates those particular species responsible for the pattern. Species with a strong relationship with the CAP axes are listed in Table 5. A positive correlation for species with the CAP axis indicates increasing numbers with fetch. Changes associated with the high fetch bays (particularly Waiwera) include increased average abundances of the echinoderms *Stegnaster inflatus* and *Patiriella regularis*, the sponge *Tethya*, the algae *Zonaria* and *Ecklonia*, and the gastropods *Haustrum*, *Cantharidus* and *Micrelenchus* sp. Species positively correlated with the low fetch bays (particularly Manly) include the algae *C. plumosum*, *Cystophora*, *Hormosira* and *Sargassum sinclairii*, solitary ascidians and the gastropods *C. virgata*, *Turbo* and *Maoricolpus roseus*.
Table 3
PERMANOVA results from analysis of the Bray-Curtis similarities of log(\(x+1\)) transformed count and cover data lumped at the year by site level. %Var is the percentage of variation explained by each factor. Note: the same analysis on presence/absence transformed data showed the same pattern of significance. Fe = Fetch, Ye = Year, Ba = Bay.

<table>
<thead>
<tr>
<th>Source</th>
<th>df</th>
<th>P(perm)</th>
<th>% Var.</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>COUNT DATA</strong></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Fetch (Fe)</td>
<td>1</td>
<td>0.001</td>
<td>7.5</td>
</tr>
<tr>
<td>Year (Ye)</td>
<td>9</td>
<td>0.001</td>
<td>12.7</td>
</tr>
<tr>
<td>Fe x Ye*</td>
<td>9</td>
<td>0.015*</td>
<td>3.1</td>
</tr>
<tr>
<td>Res</td>
<td>280</td>
<td></td>
<td>25.4</td>
</tr>
<tr>
<td>Total</td>
<td>299</td>
<td></td>
<td></td>
</tr>
<tr>
<td><strong>COVER DATA</strong></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Fetch (Fe)</td>
<td>1</td>
<td>0.001</td>
<td>5</td>
</tr>
<tr>
<td>Year (Ye)</td>
<td>9</td>
<td>0.001</td>
<td>13.9</td>
</tr>
<tr>
<td>Fe x Ye</td>
<td>9</td>
<td>0.693</td>
<td>1.3</td>
</tr>
<tr>
<td>Res</td>
<td>280</td>
<td></td>
<td>22.5</td>
</tr>
<tr>
<td>Total</td>
<td>299</td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

* Presence absence transformed data p-value = 0.913

Figure 2
Plot of the CAP axis relating biotic count data to bay-level fetch (exposure). CAP analysis used Bray-Curtis, log(\(x+1\)) transformed data. Numbers refer to years (99 = 1999, 0 = 2000, ..., 8 = 2008). Each point represents the average bay value for a year (i.e. 7 quadrats x 5 sites = 35 quadrats). Note: some values have been jittered along the x-axis for clarity.
Table 4
ANOSIM $R$-statistic and test for bay effects and Mantel test examining seriation (along a fetch gradient using Spearman’s $\rho$). Analyses were done separately for each year and used Bray-Curtis dissimilarities on log($x+1$) transformed data.

<table>
<thead>
<tr>
<th>Year</th>
<th>Bay Effects $R$</th>
<th>p-value</th>
<th>Spatial seriation (Fetch) $\rho$</th>
<th>p-value</th>
</tr>
</thead>
<tbody>
<tr>
<td>1999</td>
<td>0.176</td>
<td>1.9%</td>
<td>0.118</td>
<td>7.2%</td>
</tr>
<tr>
<td>2000</td>
<td>0.065</td>
<td>14.5%</td>
<td>0.086</td>
<td>15.6%</td>
</tr>
<tr>
<td>2001</td>
<td>0.202</td>
<td>0.7%</td>
<td>0.123</td>
<td>4.4%</td>
</tr>
<tr>
<td>2002</td>
<td>0.293</td>
<td>0.2%</td>
<td>0.282</td>
<td>0.1%</td>
</tr>
<tr>
<td>2003</td>
<td>0.212</td>
<td>0.2%</td>
<td>0.187</td>
<td>4.3%</td>
</tr>
<tr>
<td>2004</td>
<td>0.483</td>
<td>0.1%</td>
<td>0.239</td>
<td>0.2%</td>
</tr>
<tr>
<td>2005</td>
<td>0.401</td>
<td>0.1%</td>
<td>0.359</td>
<td>0.1%</td>
</tr>
<tr>
<td>2006</td>
<td>0.281</td>
<td>0.1%</td>
<td>0.373</td>
<td>0.1%</td>
</tr>
<tr>
<td>2007</td>
<td>0.365</td>
<td>0.1%</td>
<td>0.419</td>
<td>0.1%</td>
</tr>
<tr>
<td>2008</td>
<td>0.366</td>
<td>0.1%</td>
<td>0.269</td>
<td>0.2%</td>
</tr>
</tbody>
</table>

Table 5
Pearson correlation coefficients of individual species with the CAP axis for fetch (Figure 2). Only those correlations with an absolute value $>0.4$ are shown.

<table>
<thead>
<tr>
<th>Positive correlation with CAP axis</th>
<th>$\rho$</th>
<th>Negative correlation with CAP axis</th>
<th>$\rho$</th>
</tr>
</thead>
<tbody>
<tr>
<td>Stegnaster inflatus</td>
<td>0.657</td>
<td>Carpophyllum plumosum</td>
<td>-0.657</td>
</tr>
<tr>
<td>Zonaria turneriana</td>
<td>0.629</td>
<td>Cominella virgata</td>
<td>-0.588</td>
</tr>
<tr>
<td>Tethya burtoni</td>
<td>0.606</td>
<td>Cystophora sp.</td>
<td>-0.467</td>
</tr>
<tr>
<td>Patiriella regularis</td>
<td>0.573</td>
<td>Solitary ascidians</td>
<td>-0.456</td>
</tr>
<tr>
<td>Cantharidus purpureus</td>
<td>0.508</td>
<td>Turbo smaragdus</td>
<td>-0.446</td>
</tr>
<tr>
<td>Hastrum haustorium</td>
<td>0.504</td>
<td>Maoricolpus roseus</td>
<td>-0.420</td>
</tr>
<tr>
<td>Ecklonia radiata</td>
<td>0.463</td>
<td>Hormosira banksii</td>
<td>-0.418</td>
</tr>
<tr>
<td>Micrelenchus sp.</td>
<td>0.408</td>
<td>Sargassum sinclairii</td>
<td>-0.407</td>
</tr>
</tbody>
</table>
4.2 Assessing changes through time

4.2.1 Temporal patterns in community composition

Change over time was examined for significant differences in (i) dispersion and (ii) directional change (seriation) at the bay level.

Temporal dispersion

There was no evidence that bays were significantly different in their temporal dispersion (temporal variability) for either count or cover data (PERMDISP, \( p > 0.2 \)) (consider the variability of bay centroids (averaged for each year) shown in Figure 3 and Figure 4).

Temporal seriation

All bays showed significant evidence of directional change over time (i.e. temporal seriation) for count data, and Manly and Long Bay also showed seriation for cover data. Bay centroids tracked over time for count and cover data are shown in the MDS ordination plots (Figure 3 and Figure 4 respectively). Both MDS ordinations have high stress and bay variability over time is relatively high (indicating the plot is not capturing all the variability in the data cloud). Overall trends across LBMMMP are better seen when examining the ‘main-effects plots’ (Figure 5 and Figure 6). These figures show Bray-Curtis dissimilarities among main effects centroids (i.e. 6 bays (averaged across all times) + 10 time points (averaged across all bays)) for both count and cover data (Table 6). The position of bays in the main effects plots generally corroborates with the classification tree and fetch gradient. Waiwera and Manly are distinct at different ends of a roughly vertical axis for both count and cover data. Both figures also show a yearly gradient approximately positioned along the x-axis (left to right).

Table 6
Results of Mantel’s test examining seriation in LBMMMP bays (Spearman’s \( \rho \) and associated \( p \)-value). Analyses used Bray-Curtis dissimilarities of log \((x+1)\) transformed bay x year centroids for count and cover.

<table>
<thead>
<tr>
<th>Bay</th>
<th>Cover data</th>
<th>Count data</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>( p )-value</td>
<td>Rho (( \rho ))</td>
</tr>
<tr>
<td>Waiwera</td>
<td>0.003</td>
<td>0.65</td>
</tr>
<tr>
<td>Stanmore</td>
<td>0.002</td>
<td>0.65</td>
</tr>
<tr>
<td>Manly</td>
<td>0.001</td>
<td>0.80</td>
</tr>
<tr>
<td>Long Bay</td>
<td>0.001</td>
<td>0.88</td>
</tr>
<tr>
<td>Torbay</td>
<td>0.003</td>
<td>0.70</td>
</tr>
<tr>
<td>Campbells Bay</td>
<td>0.002</td>
<td>0.53</td>
</tr>
</tbody>
</table>

The associated biplots identify species abundance and cover variables (averaged across the whole LBMMMP) that are correlated with the change, including: increased abundance of \( C. flexuosum \), \( Australostichopus \) and \( Tethya \) and decreased abundance
of *Dicathais orbita* and *C. maschalocarpum* (Figure 5). The percentage cover of shell, *Chaetopterus* sp., solitary ascidians, *C. flexuosum* and the sponge *Ancorina* sp. have all increased over time. In contrast, cover of *C. maschalocarpum* and coralline turfing algae have decreased over time (Figure 6).
Figure 3
MDS plot tracking bay centroids of count data through time (using Bray-Curtis dissimilarities and \( \log(x+1) \) transformation). Numbers indicate the sample year (99=1999, .., 8 =2008).

Figure 4
MDS plot tracking bay centroids of cover data through time (using Bray-Curtis dissimilarities and \( \log(x+1) \) transformation). Numbers indicate the sample year (99=1999, .., 8 =2008)
Figure 5
MDS plot showing the bay and year centroids using count data (with Bray-Curtis dissimilarities and log(x+1) transformation). Numbers indicate sample years (99 = 1999, .., 8 = 2008). The unit circle (radius =1) below the MDS plot shows those species correlated with an MDS axis (Pearson correlation >0.75).

Figure 6
MDS plot showing the bay and year centroids using cover data (Bray-Curtis dissimilarities and log(x+1) transformed data). Numbers indicate sample years (99 = 1999, .., 8 = 2008). The unit circle (radius =1) below the MDS plot shows those species correlated with an MDS axis (Pearson correlation >0.75).
4.2.2 Temporal patterns in size frequency distributions

Of the eight taxa examined for population size structure changes, only *Cantharidus* showed no evidence of differences between bays or changes over time. *C. plumosum* changed over time, but had a similar size distribution in all bays. In contrast, the size distribution of *Cystophora* sp. differs between bays, but has remained constant over time (Table 7).

The size distribution of *C. maschalocarpum*, *C. flexuosum* and *Ecklonia, Turbo* and *Trochus* varied between bays and sampled years (Table 7) – the size frequency for these taxa are shown in Figure 7 - Figure 11. Although the typical (median) length *C. maschalocarpum* remains relatively stable at all bays, there has been a marked decline in (i) the proportion of smaller algae (less than 30 cm in length), and (ii) total abundance since 2004 (Figure 7). In contrast, *C. flexuosum* abundance has increased at all bays in recent years (with increased numbers at the four northernmost bays since 2003). Most bays show an increase in all size classes of *C. flexuosum*, but Waiwera had a particularly large proportion of recruits in the 2008 survey (Figure 8).

*Ecklonia* stipe length generally increased over time at the northern bays. Stanmore, in particular, had a marked increase of larger plants in 2008. No distinctive patterns in size frequency were apparent in the size frequency ordinations of *Turbo* or *Trochus* (Appendix G).

<table>
<thead>
<tr>
<th>Taxa</th>
<th>Bay</th>
<th>Year</th>
</tr>
</thead>
<tbody>
<tr>
<td><em>Carpophyllum maschalocarpum</em></td>
<td>0.001</td>
<td>0.001</td>
</tr>
<tr>
<td><em>Carpophyllum flexuosum</em></td>
<td>0.001</td>
<td>0.001</td>
</tr>
<tr>
<td><em>Carpophyllum plumosum</em></td>
<td>0.234</td>
<td>0.033</td>
</tr>
<tr>
<td><em>Ecklonia radiata</em></td>
<td>0.001</td>
<td>0.001</td>
</tr>
<tr>
<td><em>Cystophora</em> sp.</td>
<td>0.033</td>
<td>0.168</td>
</tr>
<tr>
<td><em>Turbo smaragdus</em></td>
<td>0.001</td>
<td>0.001</td>
</tr>
<tr>
<td><em>Trochus viridus</em></td>
<td>0.026</td>
<td>0.001</td>
</tr>
<tr>
<td><em>Cantharidus purpureus</em></td>
<td>0.158</td>
<td>0.201</td>
</tr>
</tbody>
</table>
The size frequency distribution of *Carpophyllum maschalocarpum*. The red dashed line indicates the median length.
Figure 8
The size frequency distribution of *Carpophyllum flexuosum*. The red dashed line indicates the median length.
Figure 9
The size frequency distribution of *Ecklonia radiata* stipe length. The red dashed line indicates the median length.
Figure 10
The size frequency distribution of *Turbo smaragdus*. The red dashed line indicates the median length.
Figure 11
The size frequency distribution of *Trochus viridus*. The red dashed line indicates the median length.
4.2.3 Temporal patterns in univariate variables

4.2.3.1 Temporal patterns in count variables

The six most abundant species in the LBMMP constituted over 80% of all counted individuals, so examining the temporal trends of these species should give a good representation of the major trends. The most abundant species counted in the LBMMP were (in decreasing order of abundance): *C. maschalocarpum*, *Turbo*, *Zonaria*, *C. plumosum*, *C. flexuosum* and *Ecklonia*.

Univariate indices

The results from the univariate GLMM analyses examining changes over time for the various indices are summarized in Table 8. Confidence intervals for total abundance data are expressed as a percentage change in abundance per year (i.e. a multiplicative [proportional] change); in contrast, confidence intervals for algal biomass, diversity and evenness indices are expressed as an additive change.

<table>
<thead>
<tr>
<th>Index</th>
<th>Campbells</th>
<th>Torbay</th>
<th>Long Bay</th>
<th>Manly</th>
<th>Stanmore</th>
<th>Waiwera</th>
</tr>
</thead>
<tbody>
<tr>
<td>Total abundance</td>
<td>(5-10)%</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>(3-9)%</td>
</tr>
<tr>
<td>Algal biomass</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Diversity (count)</td>
<td>(0.3-0.6)</td>
<td>(0.4-0.8)</td>
<td>(0.2-0.6)</td>
<td>(0.2-0.7)</td>
<td>(0.7)</td>
<td>(0.1-0.3)</td>
</tr>
<tr>
<td>Evenness (count)</td>
<td>(0-0.03)</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

Changes over time of the following indices for each of the univariate indices shown in Table 8 are shown in Figure 12 - Figure 15. These figures show a lattice plot with points corresponding to averaged site abundance. For clarity, site standard errors were not shown. A black least squares regression line and a red local regression smoother were fit to each site-panel as an indication of trend over time.

Salient results include evidence that (after accounting for depth and ENSO effects):

- Total abundance increased slightly at Campbells and Waiwera (by between 5 - 10% and 3 – 9% per year respectively).
- Diversity for counted biota increased over time at all bays. From 1999 - 2008, the average diversity per site has increased by between 2 - 8 species.

Note: Despite some major changes in algal composition, there was no evidence of a change (over time) in algal biomass in any bay in the LBMMP.

\[\text{Site level plots tracking these species (and crustose coralline algae cover) are shown in section 7.8, Appendix H.}\]
Figure 12
Change in total abundance per site. Panel points represent the average site abundance. A black least squares (LS) regression line and a red local regression smoother are fitted to each panel. The small orange stripe in each title panel represents the bay’s northern position (left side = South, right-side = North).

Figure 13
Algal biomass – figure details are explained in the caption of Figure 12.
Figure 14
Average site diversity – figure details are explained in the caption of Figure 12.

Figure 15
Site evenness – figure details are explained in the caption of Figure 12.
Macroalgal abundance

Changes in macroalgal abundance over time are summarized in Table 9. Confidence intervals are expressed as a percentage change in abundance per year (i.e. a multiplicative [proportional] change).

Table 9
GLMM model results showing the change in macro-algal abundance over time (after accounting for climate and depth effects). Red font indicates a linear decrease over time; \( NL \) indicates a non-linear change over time (the non-linearity can be visualized in Figure 16 - Figure 20). Values indicate the 95% confidence interval of the estimated change per year.

<table>
<thead>
<tr>
<th>Counted biota</th>
<th>Campbells</th>
<th>Torbay</th>
<th>Long Bay</th>
<th>Manly</th>
<th>Stanmore</th>
<th>Waiwera</th>
</tr>
</thead>
<tbody>
<tr>
<td>C. maschalocarpum</td>
<td>-(1-8)% NL</td>
<td>-4-14% NL</td>
<td>NL</td>
<td>NL</td>
<td>NL</td>
<td>NL</td>
</tr>
<tr>
<td>Zonaria turneriana</td>
<td>(10-18)%</td>
<td>(11-19)%</td>
<td>(6-15)% NL</td>
<td>NL</td>
<td>NL</td>
<td>NL</td>
</tr>
<tr>
<td>C. plumosum</td>
<td>-(22-45)%</td>
<td>-(7-53)%</td>
<td>NL</td>
<td>NL</td>
<td>NL</td>
<td>NL</td>
</tr>
<tr>
<td>C. flexuosum</td>
<td>(17-33)%</td>
<td>(18-51)%</td>
<td>(3-23)%</td>
<td>(16-40)%</td>
<td>(4-20)%</td>
<td>NL</td>
</tr>
<tr>
<td>Ecklonia radiata</td>
<td>(1-12)%</td>
<td>(1-16)%</td>
<td>-(3-13)%</td>
<td>NL</td>
<td>NL</td>
<td>NL</td>
</tr>
<tr>
<td>Cystophora sp.</td>
<td>(12-27)% NL</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

Changes in abundance over time were plotted for those species mentioned above (Figure 16 - Figure 20). Salient points include:

- **C. maschalocarpum** abundance decreased markedly over time at all bays. The abundance at northern bays (Manly, Stanmore and Waiwera) and Torbay was relatively stable until 2002, but declined rapidly between 2002 and 2004. Since 2004, **C. maschalocarpum** abundance has been relatively stable. Campbells and Long Bay had a stable but marked decline in abundance; we estimate that they have declined by 1 - 8% and 4 - 14% per year respectively.

- **Zonaria** abundance increased at all bays except Manly and Stanmore. At all three northern bays the population increased between 1999 and 2004, followed by a decline until 2006. After 2006, the Stanmore population of **Zonaria** continued to decline, while the populations at Manly and Waiwera stabilized and increased respectively. The **Zonaria** population at the three southern sites increased markedly over time. We estimate, on average, that the population of **Zonaria** has increased by between 10 - 18% per year at Campbells (with the 2008 survey showing signs of a larger proportional increase), 11 - 19% at Torbay and 6 - 15% at Long Bay.

- **C. plumosum** abundance declined at the two northernmost bays. Stanmore and Waiwera decreased markedly (we estimate an average decline of between 22 - 45% and 7 - 53% per year respectively), but they had relatively low initial abundance, so, in absolute terms, their decline was small.

- **C. flexuosum** abundance increased markedly at all bays except Long Bay. The three northern bays show the greatest increase occurred between 2002 and
2005, while the southern bays increased most markedly since 2005. Torbay shows the largest proportional increase (we estimate between 18 - 51% per year), followed by Stanmore (16 – 40%), Campbells (17 - 33%), Manly (3 – 23%) and Waiwera (4 – 20%).

- *Ecklonia* abundance has increased at Campbells, Long Bay and Waiwera (with most change taking place mainly prior to the 2005 sampling), but decreased at Manly (by 3 -13%). We estimate that, on average *Ecklonia* has increased at Campbells by 1 – 12%, Long Bay by 1 – 16% and Waiwera by between 5 – 12% (per year).

- *Cystophora* abundance appears to be both spatially and temporally patchy. Some sites within a bay have consistently had zero counts, while other sites change from low abundance to relatively large numbers in the following year. Manly has shown the most spatial and temporal consistency of all the LBMMP bays (we estimate an increase of between 12 – 27% per year). In contrast, increases in abundance at Waiwera appeared at only a couple of sites. *Cystophora* abundance at Stanmore peaked at 2004 and has declined since then.

---

**Figure 16**

*Carpophyllum maschalocarpum* – figure details are explained in the caption of Figure 12.
Figure 17
Zonaria turneriana – figure details are explained in the caption of Figure 12.

Figure 18
Carpophyllum plumosum – figure details are explained in the caption of Figure 12.
Figure 19
*Carpophyllum flexuosum* – figure details are explained in the caption of Figure 12.

![Graph showing the average abundance per site for *Carpophyllum flexuosum* across different locations and years.](image)

Figure 20
*Ecklonia radiata* – figure details are explained in the caption of Figure 12.

![Graph showing the average abundance per site for *Ecklonia radiata* across different locations and years.](image)
Changes in macroinvertebrate abundance

The change in the most numerous macroinvertebrates are shown in Table 10. In the previous LBMMMP report, Anderson et al (2005) noted (i) increasing abundances of: Australostichopus, Cookia sulcata, Patiriella regularis, Coscinasterias muricata and Buccinulum lineum, and (ii) decreasing abundances of Dicathais orbita, Sargassum sinclairii and Cominella virgata. Although changes in abundance for all these species have not been plotted (these taxa are comparatively rare), we found that the abundances of these species have stabilized since 2005.
Table 10: GLMM model results showing the change in invertebrate abundance over time (after accounting for climate and depth effects). Red font indicates a linear decrease over time; NL indicates a non-linear change over time (the non-linearity can be visualized in). Values indicate the 95% confidence interval of the estimated change per year.

<table>
<thead>
<tr>
<th>Counted biota</th>
<th>Campbells</th>
<th>Torbay</th>
<th>Long Bay</th>
<th>Manly</th>
<th>Stanmore</th>
<th>Waiwera</th>
</tr>
</thead>
<tbody>
<tr>
<td>Turbo smaragdus</td>
<td>(3-20)%</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Trochus viridus</td>
<td>(3-15)%</td>
<td>-(3-14)%</td>
<td>-(2-12)%</td>
<td>(1-19)%</td>
<td>(15-41)%</td>
<td></td>
</tr>
<tr>
<td>Cantharidus purpureus</td>
<td>(14-32)%</td>
<td>NL</td>
<td></td>
<td></td>
<td>(17-34)%</td>
<td></td>
</tr>
<tr>
<td>Solitary ascidians</td>
<td>(11-34)%</td>
<td>(13-42)%</td>
<td>(17-40)%</td>
<td>NL</td>
<td>(6-22)%</td>
<td>(3-29)%</td>
</tr>
<tr>
<td>Tethya burtoni</td>
<td>(23-92)%</td>
<td>(10-45%)</td>
<td>(2-52)%</td>
<td>NL</td>
<td>(30-61)%</td>
<td></td>
</tr>
<tr>
<td>Evechinus chloroticus</td>
<td></td>
<td>-(2-15)%</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Maoricolpus roseus</td>
<td></td>
<td></td>
<td></td>
<td>NL</td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

Salient points from Table 10 include:

- **Turbo** abundance increased at Campbells (Figure 22). We estimate abundance at Campbells has increased between 3 - 20% per year.
- **Trochus** abundance decreased at Torbay, Long Bay and Manly (the three bays with the lowest fetch in the LBMMP) by between 3 – 14%, 2 – 12% and 12 - 23% per year respectively. In contrast, abundance increased at Campbells, Stanmore and Waiwera by between 3 - 15%, 1 – 19% and 15 - 41% respectively. Waiwera site 4, in particular, showed a marked increase - from zero individuals found in 1999, to an average of ~13 per quadrat from 2005 - 2008) (Figure 23).
- **Cantharidus** abundance has increased at Waiwera and Campbells, but remains relatively unchanged at other bays (Figure 24)
- Solitary ascidian abundance increased at all bays, but particularly at Torbay, Long Bay and Manly (i.e. those bays with the least fetch) (Figure 25). In particular, solitary ascidian abundance at Manly has risen dramatically at sites 1 and 3 since 2007. We estimate that solitary ascidian abundance has increased by between 13 - 42% and 17 - 40% per year at Torbay and Long Bay, and between 11 – 34%, 6 – 22% and 3 – 29% at Campbells, Stanmore and Waiwera respectively.
- **Tethya** has increased at all bays, put particularly large increases have been seen since 2005 in high fetch bays (i.e. Campbells, Stanmore and Waiwera) (Figure 26). Low initial abundance means that the increase in Long Bay (2 - 52%) and Torbay (10-55%) per year is not large in absolute numbers. Abundance at Campbells and Waiwera have increased between 23 -92% and 30 -61% per year.
- **Evechinus** has decreased at Long Bay. We estimate a decrease between 2 – 15% per year (Figure 27).
• *Maoricolpus* abundance is spatial and temporally patchy at most sites (Figure 28). In 2004, one site in Stanmore averaged more than 30 per quadrat, but has had no high abundance since that time. *Maoricolpus* abundance has shown increased abundance at three sites in Manly since 2002.

**Figure 22**  
*Turbo smaragdus* – figure details are explained in the caption of Figure 12.
Figure 23
*Trochus viridus* – figure details are explained in the caption of Figure 12.

![Graph showing average abundance per site for *Trochus viridus* over years from 2000 to 2008 across different sites.](image)

Figure 24
*Cantharidus purpureus* – figure details are explained in the caption of Figure 12.

![Graph showing average abundance per site for *Cantharidus purpureus* over years from 2000 to 2008 across different sites.](image)
Figure 25
Solitary ascidians – figure details are explained in the caption of Figure 12.

Figure 26
*Tethya burtoni* – figure details are explained in the caption of Figure 12.
Figure 27
_Evechinus chloroticus_ – figure details are explained in the caption of Figure 12.

![Graph showing average abundance per site for _Evechinus chloroticus_ across different sites and years.]

Figure 28
_Maoricolpus roseus_– figure details are explained in the caption of Figure 12.

![Graph showing average abundance per site for _Maoricolpus roseus_ across different sites and years.]

### Temporal patterns in cover variables

The results from the univariate GLMM analyses examining percentage cover changes over time are summarized in Table 11. Confidence intervals in cover variables are expressed as an additive change in percentage cover per year.

Changes in percentage cover over time (as well as total abundance and site evenness) are shown in Figure 29 - Figure 32.

#### Table 11

GLMM model results showing the change in cover variables over time (after accounting for climate and depth effects). Red font indicates a linear decrease over time; *NL* indicates a non-linear change over time (the non-linearity can be visualized in Figure 29 - Figure 34). Values indicate the 95% confidence interval of the estimated change per year.

<table>
<thead>
<tr>
<th>Cover variables</th>
<th>Campbells</th>
<th>Torbay</th>
<th>Long Bay</th>
<th>Manly</th>
<th>Stanmore</th>
<th>Waiwera</th>
</tr>
</thead>
<tbody>
<tr>
<td>CCA</td>
<td>(1-3)%</td>
<td>-(-0-3)%</td>
<td>NL</td>
<td>NL</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Sediment cover</td>
<td><em>NL</em></td>
<td>(2-3)%</td>
<td>NL</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Diversity (cover)</td>
<td>(0.3-0.6)</td>
<td>(0.4-0.7)</td>
<td>NL</td>
<td>NL</td>
<td>NL</td>
<td>NL</td>
</tr>
<tr>
<td>Evenness (cover)</td>
<td>(0-0.01)</td>
<td>(0-0.01)</td>
<td>(0-0.02)</td>
<td>(0-0.01)</td>
<td>(0-0.01)</td>
<td>(0-0.01)</td>
</tr>
</tbody>
</table>

Salient results include evidence that (after accounting for depth and ENSO effects):

- All bays showed a marked increase in cover diversity between 1999 and 2003. Since 2003, diversity dropped at Stanmore and Waiwera, and stabilized or increased slightly at other bays. On average, site diversity of cover variables in LBMMP has increased by between 2 – 7 species since 1999.

- Assemblage evenness increased by a small amount at all bays. We estimate that Long Bay has an average increase in the evenness index of between 0 - 0.02 units per year; other bays have increased by between 0 – 0.01 units.

- Crustose coralline algae (CCA) cover decreased at Long Bay (by between 0 – 3% per year), but increased at Campbells (by between 1 - 3% per year). CCA at Stanmore and Waiwera has increased since 2002.

- On average, sediment cover increased at Long Bay by between 2 – 3% per year. Campbells and Stanmore showed a decline in sediment cover between 1999 and 2002, followed by an increase back to original levels.