Assessing ecological patterns in Wellington south coast's nearshore rocky-reef communities for resource conservation and management

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ABSTRACT

Many coastal marine communities are increasingly affected by terrestrial and maritime human activities and growing coastal populations. Protection of coastal assets and the sustainable use of coastal resources requires knowledge of nearshore benthic community status; the environmental processes that structure and connect them; the quality, abundance, and distribution of physical habitat; essential habitat for species requiring protective measures, and the spatio-temporal scales at which these patterns and processes occur. To assess the status of Wellington South Coast's (WSC) rocky-reef assemblages prior to the enactment of the Taputeranga Marine Reserve in 2009, two annual baseline surveys were conducted during the austral summers of 2007/08 and 2008/09. These surveys evaluated the biotic and abiotic components of the assemblages in terms of diversity, abundance distribution, and size-class frequency patterns of key macro-algal and mobile macro-invertebrate species. These results were analysed to develop recommendations for best post-reserve monitoring practices, including the identification of "indicator" species for rapid yet representative field surveys to assess structural and status changes. In combination with patterns described by a previous pre-reserve baseline survey series (2000) that focussed on a reduced list of macro-algal and mobile macro-invertebrate species, this final pre-reserve survey forms the basis of a historical dataset for WSC rocky reefs that can be used for long-term monitoring of ecosystem shifts due to the new reserve and to possible changes caused by anthropogenic activity or altered natural processes.

These aims were addressed by collecting information directly at local/site scale and remotely, at the larger area scale. Dived baseline surveys quantified nearshore WSC rocky-reef epibenthic assemblages at 9 sites at depths ranging from 5-13.6m and at a mean distance from shore = 113m. A survey design that included three sites west of the pending marine reserve, three sites to the east, and three sites within the designated reserve was selected to permit later BACI analyses of post-reserve changes. Species surveyed were those commonly encountered during daylight on exposed surfaces and in accessible crevices and belonged to one of three epibenthic groups: macro-algae (48 species), mobile macro-invertebrates (36 species), and sessile macro-invertebrates (30 morphotypes). These surveys did not include epizoa or smaller, cryptic newly recruited macro-invertebrates. Sessile macro-invertebrate cover was only logged if $>0.1\%/m^2$. To gauge possible spatio-temporal patterns in primary productivity as a measure of ecosystem function, biomass and plant size were measured semi-annually (winter, summer) for dominant kelp and furoid species and for two key recreationally and commercially important mobile macro-invertebrate species (sea urchin and abalone). Predictive regression equations developed from wet weight and plant size can be used for future non-destructive estimates of local primary productivity and in trophic modelling.

Environmental variables including temperature, salinity, dissolved oxygen, and Secchi depth were monitored monthly for two years at 8 of the 9 survey sites along the 14-km coast and within the same depth range as the dives surveys (5-14m). Additional environmental factors analysed for their contribution to assemblage structure included seabed Secchi distance, fetch, hydrology, harbour proximity, seabed composition and level of urban development.

SIMPER analyses identified species contributing to spatio-temporal heterogeneity based on abundances, while analyses of biodiversity indices (α and β) derived from presence-absence data described spatial patterns in taxonomic composition. Rhodophyta and *Macrocystis pyrifera* were among the taxonomic discriminators of eastern sites. Gastropods discriminated central and eastern sites. BEST routines identified representative subsets of each group that would act as representative proxies for full assemblages in future monitoring. PERMDISP identified spatial patterns in the biomass of 10 common canopy, understory, and

turfing macro-algae and 3 mobile macro-invertebrates. Size-frequency plots described spatial and interannual patterns for common benthic species. These spatial patterns were species-specific.

Variation in seabed composition was identified by χ^2 analyses, but no clear south coast gradient was detected. The greatest variability in physical habitat composition was detected at two eastern sites. Reef orientation was primarily horizontal except at two central sites, where vertical reef dominated and rugosity values were highest. They were lowest at one eastern site and similar at all other sites. Seasonal patterns in water parameters (temperature, dissolved oxygen, salinity, and visual clarity/Secchi depth) measured monthly from 2007-2009 at eight of the nine dived baseline survey sites were analysed with generalised linear models. The contribution of these and other environmental variables to assemblage patterns were assessed using DistLM and dbRDA analyses. Environmental variables affected each of the three epibenthic groups differently. The contribution of each factor was relatively limited, with no one factor contributing more than 25% to multivariate assemblage patterns for any of these functional groups.

The larger-scale data was remotely collected by drop camera. The unit was deployed at 603 waypoints at depths ranging from ~2 - 50m for a final coverage of 2,192 hectares of WSC's nearshore seabed. Video and extracted stills were analysed to develop the first benthic habitat maps of Wellington's south coast. The semi-quantitative yet rapid SACFOR method (video) was compared with a fully quantitative, slower visual assessment method (stills) to identify possible differences in identifying canopy abundance. Map products were developed using geographical information systems software (GIS). The effects of different sampling intensities (603 vs. 450 vs. 300 vs. 150 waypoints) and image analysis method (video vs. extracted stills) on habitat and biota identification were assessed using χ^2 analyses and two-way ANOVA without replications. No differences in frequency of camera drop classifications by sampling intensity were identified. Maps presented benthic descriptions at two levels of resolution for physical habitat, major seabed class and subclass. Macro-algal canopy cover was the only biotic group that could be consistently, reliably, and repeatably identified in both video clips and extracted still frames. Understorey, turfing and encrusting taxa were less consistently identifiable and cryptic and mobile species were typically not detectable. Habitat association was identified at two levels of resolution, macro-algal canopy species and major functional group, using the R code package Adehabitat based on Manly's alpha. Geostatistical techniques identified the spatial characteristics of the macro-algal canopy layer.

This mapping data forms the basis of a legacy dataset that will assist with monitoring changes in the integrity of critical physical habitat and associated biotic cover. It has also demonstrated that representative descriptions of both biotic and abiotic benthic components can be achieved with a minimum of sampling points and by using the quicker semi-quantitative visual analysis of video. These data can also be used to ground-truth a recently-completed multi-beam acoustic survey of the area.

This work has used the approach of landscape ecology, which explains patterns in community structure, function, status and biophysical causes from a spatial perspective, to study biophysical patterns in WSC epibenthic rocky-reef communities. The work identified a high degree of spatial and temporal variation within the abiotic and biotic community within and outside of the reserve area and the limited availability of preferred habitat. The work also identified the need to include indicator species in monitoring to improve the chance of detecting impacted assemblages. These results, and the development of non-destructive sampling tools for assessing ecosystem status, are relevant locally and nationally for resource managers.

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“However much you knock at Nature’s door, she will never answer you in comprehensible words”

Ivon Turgenev, “On the Eve”

From “The Glass Harmonica”, Barbara Ninde Byfield

"There is something fascinating about science. One gets such wholesale returns of conjecture out of such a trifling investment of fact."

Mark Twain, “Life on the Mississippi”

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Chapter 1. General Introduction

1. Coastal ecosystems conservation

Conservation management principles for the marine environment have evolved from the species-based protectionist approach of developed nations to the current multi-species ecosystem approach adopted by developed and developing countries (Ray & McCormick-Ray, 2004). This shift arose from the realisation that human threats to species survival, biodiversity and habitats requires integrated scientific resource assessments in addition to political, economic, and social responses. The 1980 International Union for Conservation of Nature's (IUCN) World Conservation Strategy outlined three guiding principles for governments to incorporate into marine resource conservation programmes: 1) the protection of ecosystem function (e.g. primary and secondary productivity, sustained trophic structure); 2) preservation of genetic diversity; and 3) sustainable use of species and ecosystems (IUCN, 1980). The Rio Convention on Biological Diversity (1992) supported this integrated approach by implementing legislation requiring signatories to avoid practices that threaten conservation values beyond their maritime jurisdiction (Jeffries, 2006; Ray & McCormick-Ray, 2004).

1.1 Marine reserves in conservation: general overview

An increasingly common conservation tool for protecting biodiversity, unique habitat, and ensuring ecosystem sustainability is the spatial or temporal restriction of activities in marine areas. The designations and level of protection assigned vary internationally and include marine protected areas, marine parks, areas of traditional use, equipment protection zones, and sanctuaries. Activities can be restricted temporally or spatially through the implementation of multi-use or multi-zone regulations. Categories of restricted use include scientific research, tourism, education, and subsistence, traditional, recreational, or commercial harvesting. The most stringent classification is the marine reserve, in which no extractions are allowed. Over 2,000 protected areas have been designated globally, but these constitute <2% of the world's oceans (Ray, 1991; New Zealand Parliament, 1996; Allison *et al.*, 1998; Halpern, 2003).

Reserves can protect spawning and nursery grounds, biological and physical diversity hotspots, and create spatial buffers against multiple and often interactive anthropogenic effects. Direct benefits can include increased abundances and the re-establishment of the natural range of age and size classes for depleted stocks. These shifts in population and assemblage structures can in turn lead to reproductive success, greater genetic diversity, increased resilience, population growth, shifts in trophic structure and productivity, and spillover of reproductive propagules and organisms to nonreserve areas. Reserves can also serve as

important control areas in which naturally occurring biotic and abiotic patterns and processes can provide benchmarks against which the impacts of human activity can be assessed. Understanding these processes helps to implement their protection elsewhere, and also allows marine resource managements to implement controls on the human activities that might affect these processes (Allison et al., 1998; Ballantine & Langlois, 2008; Halpern, 2003).

Reserve effects have been demonstrated but the literature does not identify neatly classified results. Reserve effects as identified by biological measures (i.e. shifts in abundance, size class, sex ratios) have been observed to occur within a few years of implementation (Edgar & Barrett, 1999; Halpern & Warner, 2002), but effects have also been reported as slow to appear, inconsistent, occurring in stages, and varying by taxonomic group (Stoms et al., 2005). A literature review found that reserve size was not related to output measures (Stewart et al., 2008), while a monitoring survey of Tasmanian reserves found that size did contribute to increases in biological attributes (Edgar & Barrett, 1999). Results also vary within and among study groups: Stoms' (2005) review of temperate and tropical reserve monitoring studies found that density increases varied from 23-196%, biomass from 20-422%, and species richness from 10-130% relative to external sites.

Factors obscuring the detection of reserve effects include the complexity and time delay of trophic responses, the confounding effect of variable habitat quality, lack of baseline data, inadequate sampling design or statistical power, different life cycles of monitored species, and monitoring studies on inappropriate species. Also noted in literature reviews of reserve monitoring studies was a lack of baseline data and information on biomass, species richness, macro-algae, and macro-invertebrates (Ballantine & Langlois, 2008; Cole et al., 1990; Stewart et al., 2008). Monitoring studies tend to focus on impacted fish and mobile macro-invertebrate species rather than a broader range of taxa within each group (Ballantine & Langlois, 2008; Langlois & Ballantine, 2005). When a lack of reserve-related changes in biodiversity and other outcome measures were noted, this was attributed less to a lack of reserve impact and more to the lack of appropriately-designed coastal management programs, or to displaced fishing activity (Hilborn et al., 2004).

1.2 Coastal marine conservation in New Zealand

New Zealand's maritime estate, comprising its territorial waters, exclusive economic zone, and the extended continental shelf seas, covers more than six million km² and extends from the subtropics to the sub-Antarctic islands (29°S – 59°S). This area's biodiversity is estimated to be greater than similar areas in the northern hemisphere, with endemic species comprising approximately 51% of the

estimated 17,135 marine species identified to date in New Zealand (Costello et al., 2010). Currently, less than 1% of New Zealand's maritime estate has been placed into the marine protected system detailed below (Craig et al., 2000).

New Zealand's current coastal marine management practices incorporate the IUCN's mandates to ensure resource sustainability, preserve ecosystem health and maintain biodiversity. As the first country to establish the world's first fisheries quota management system in 1986, New Zealand has identified its commitment to the long-term sustainable use of marine resources. Government acts, statements, and policies have outlined specific scientific, ecological, economic, social, cultural, and historical goals relating to coastal resource use. Current resource protection policies acknowledge the interactive biological and environmental processes that define the nearshore marine environment at local, regional, and national scales. They also address the need to remediate degraded coastal habitats (The Marine Mammals Protection Act, 1978; The Conservation Act, 1987; The Resource Management Act, 1991; The Fisheries Act, 1996; The New Zealand Biodiversity Strategy, 2000; The New Zealand Coastal Policy Statement, 2010) (Craig et al., 2000; Department of Conservation, 2010). This integrative resource management approach has led to the development of a Marine Environment Classification scheme based on biotic and abiotic characteristics of the country's coastal habitats (Department of Conservation and Ministry of Fisheries, 2005; Beaumont et al., 2008; Snelder et al., 2007) and the identification of coastal marine biogeographic provinces based on biotic distribution patterns and environmental characteristics (Shears et al., 2008).

Ecological threats to New Zealand's coastal ecosystems include the introduction of alien species through vectors including currents, hulls and ballast water of international and local vessels from infected areas. Currently, ~177 marine species are now established locally and have actual or potential impact(s) on endemic species and community structure and function through resource competition and predation (Gordon et al., 2010). Other identified risks are posed by recreational and commercial fishing practices, which have led to reduced biological and physical diversity through stock and bycatch depletion and fisheries-related habitat degradation (trawling and dredging). An indirect effect includes trophic shifts due to altered stock abundances (Thrush & Dayton, 2002; Thrush et al., 2002). Other effects derive from the nearshore environment's vulnerability to anthropogenic activities and environmental events in both the terrestrial and marine realms. Physical and chemical household, industrial, agricultural, light, noise and thermal pollution; urbanisation; agricultural, industrial, mining and recreation activities; watershed and estuary modification; altered sediment

transport patterns; maritime shipping; atypical weather and storm patterns; climate-change impacts (ocean acidification, rising sea levels and temperatures) all affect nearshore biotopes by altering physical habitat availability and quality, water chemistry, life cycle patterns, resource use, resilience, and biotic and abiotic interactions (Gordon et al., 2010; Ray, 1996).

In New Zealand a lack of knowledge about coastal ecosystem structure and function also been identified as a significant problem in identifying and mitigating risk impacts (Anderson, 2002; Weston, 2012). These knowledge gaps include limited or no taxonomic guides or specialists, limited knowledge of the presence, range, and effects of alien species, a limited understanding of the nature, scale and effects of climate change and other anthropogenic effects on coastal ecosystems composition, structure, physical and biological diversity, an incomplete marine reserves network, and limited knowledge of species' life cycles and distribution. Also, information about the biological attributes of one taxon, when available, cannot always be used to explain patterns in the attributes of other taxa (Ballantine & Langlois, 2008; Boffa Miskell Limited, 2001; Craig et al., 2000; Hayden et al., 2009; Snelder et al., 2007; Weston, 2012).

A significant obstacle to a cohesive national marine reserve monitoring program capable of identifying and tracking patterns, and biotic and abiotic impacts, has been the lack of continuity between available baseline surveys when they exist and subsequent monitoring surveys. Also lacking are a national reserve monitoring database and standardised field protocols for reserve baseline and monitoring surveys. Also, patchy collection and availability of metadata makes quality assessment of data difficult (McCrone, 2001). Coastal management and policy development would also benefit from surveys on community structure and processes and status for unprotected nearshore areas as well (Oliver & Milne, 2012; Schiel & Hickford, 2001).

New Zealand's 1971 Marine Reserves Act defined the purpose of reserves to be for "management of areas of the sea and foreshore...for the purpose of preserving them in the natural state ... for scientific study" (New Zealand Parliament, 1996). This restrictive mandate has recently been expanded to include ecological targets by a new policy that calls for future reserve areas to protect "representative... outstanding and rare" ecological characteristics (Marine Protected Areas: Policy and Implementation Plan) (Department of Conservation & Ministry of Fisheries, 2005). How these qualities are to be defined is not specified. Protection measures include marine reserves, marine parks, marine protected areas, fisheries and seasonal closures, marine mammal sanctuaries, cable protection zones, and two customary protection tools used by *iwi* (tribe or people) and *hapu* (sub-tribe or

clan): *mātaitai* and *taiapure*. To date, ~7.6% of territorial waters have reserve status, 99% of which is held in two marine reserves, the Kermadecs and Auckland Islands. The first marine reserve, the Cape Rodney to Okakari Point reserve, was created in 1975 and currently there are 34 reserves ranging in size from ~93 to ~745,000 hectares, with over half implemented after 2000. Offshore, 17 benthic protection areas limit the use of deep-sea trawling and dredging methods for harvesting benthic species (www.mpi.govt.nz).

The open and interconnected nature of the marine environment requires protection of current-driven corridors along which reproductive propagules and organisms travel. To protect resilience through genetic diversity, these environmental and biological links must be identified and incorporated into the planning marine reserve networks (Bell & Okamura, 2005). To this end, the New Zealand Biodiversity Initiative has set a goal of placing 10% of territorial waters into a network of protected marine areas (Department of Conservation & Ministry of Fisheries, 2005).

The south coast's Taputeranga Marine Reserve was gazetted in September 2009 and protects 855 hectares. Local communities petitioned for its creation on the basis of its "recreational, commercial, and education opportunities" in addition to its primary role as an area for scientific research (Gardner et al., 2008). The area's appeal to the public and to the scientific community derives from the site's unique environmental characteristics and structurally complex hard-bottom habitat, which support diverse communities of macro-algae, sessile and mobile macro-invertebrates, and fish (see following section for a complete site description). Its location between two nearby reserves, Kapiti Marine Reserve and the south island's Long Island Marine Reserve, allows it to serve as a biological corridor linking the two.

1.3 Assessing temperate benthic communities for conservation practices

Conservation of kelp forest ecosystems requires knowledge of local spatio-temporal patterns and their interactions with the environment. Temperate rocky-reef communities are mosaics formed by natural variation in distribution, structure, biological and environmental processes, and resource availability. Forests are among the most productive of marine and terrestrial systems and their vertebrate and invertebrate components serve as spawning grounds, nurseries, and refuge and structure the biotic and abiotic components of reefs through biological and physical processes (Andrew & Mapstone, 1987; Bell, 2008; Dayton et al., 1998; Kennelly, 1987; Valiela, 1995). Baseline surveys coupled with targeted monitoring programs are essential tools for the identification and assessment of anthropogenic impacts and their influence on these marine systems. This information can then be

used to develop appropriate management responses (Baskett et al., 2007; McCrone, 2001). However, the evaluation of pre- and post-reserve impacts on ecosystems is often hindered by factors including a lack of pre-reserve baseline survey, surveys focusing on groups or taxa that do not necessarily represent the full assemblage's multivariate structure, and survey designs unsuitable for the single impact site created by a reserve and by inadequate controls (Kingsford & Battershill, 1998; Underwood, 1994).

Well-designed field surveys provide information on spatial and temporal patterns in biotope structure, distribution, and range, as well as information on the geophysical variables that structure the assemblages. Because underwater surveys are costly and constrained by the limits of SCUBA and dive site access, visibility and sea conditions, dived survey methods that maximise the amount of information collected are desirable. Multivariate analyses of baseline survey data can identify representative subgroups of more complete species lists in order to expedite subsequent monitoring. Such analyses can also assess the adequacy of using functional groups and morphotypes over taxonomic identities when surveying less well identified taxa or surveying with volunteers (Bates & DeWreede, 2007; Bell, 2007; Bell et al., 2006; Obrist & Duelli, 2010; Steneck & Dethier, 1994).

Biological attributes assessed include: abundance (density, percent cover, counts); productivity and ecosystem processes (biomass), population structure (size-frequency distributions); and taxonomic composition (diversity). Diversity comprises two components, species richness and equitability, and is defined at several spatial scales: point, alpha, beta, gamma, and omega, or phylogenetic, diversity. Point diversity describes the taxonomic variation of a micro-environment but this is more often assessed at the level of α -diversity, or local habitat. β -diversity describes the turnover in taxonomic composition between sites along a spatial gradient, while γ -, or landscape diversity, identifies the taxonomic characteristics at the regional level. Alpha-diversity patterns are driven by local ecological processes; β -diversity by these and by physical habitat differences, while evolutionary processes drive γ -diversity (Whittaker, 1972; Gray, 1997). Diversity indices have been used to track community changes in response to physical stressors, large-scale community turnover, and the effect of habitat distribution in structuring soft-sediment assemblages (Bates & DeWreede, 2007; Bates et al., 2005; Ellingsen et al., 2005; Hewitt & Funnell, 2005).

Marine habitat mapping identifies spatial patterns in benthic communities, physical habitat use, and the relationships between communities, environmental processes, and human activities. Biotic and abiotic data sets can be used in models exploring the response of benthic biotopes to anthropogenic impacts. These models

are used in ecosystem-based management, a conservation approach based on sustainability, representativeness, ecological processes, connectivity, and long-term protection (Cogan et al., 2009). Baseline knowledge of environmental processes, their relationships to biological patterns, the scale at which the two interact, and how seabed communities respond to natural and anthropogenic changes helps managers to incorporate spatial considerations into management strategies (Davies et al., 1997; Pickrill & Todd, 2003).

Small-scale patterns in geomorphology and associated biota can be identified directly through dived video surveys but larger, deeper areas are surveyed through remote sensing methods that include acoustic surveys, satellite imagery, and aerial photography. Acoustic surveys are coordinated by video and still camera surveys, but drop-cameras can be used as a stand-alone technique. Output is typically collected, processed, analysed, stored, and presented within the framework of a geographical information system (Davies et al., 2001). Map products such as thematic and biological valuation maps are valuable data communication tools to lay and technical audiences (Derosus et al., 2007)

Relationships between remotely collected geophysical and biological data can be used to identify physical surrogates for biological communities, processes, diversity hot-spots. These surrogates facilitate the rapid collection of data over large areas. Geophysical data collected includes seabed morphology, rugosity, slope, depth, water quality, productivity, temperature. However, gradients in geophysical variables can obscure the identification of biotic-abiotic relationships. This was identified as an issue during a side-scan survey of Te Whanganui-a-Hei Marine Reserve and resulted in a recommendation for higher-resolution sampling to resolve the issue (Hewitt et al., 2004). Other uses of seabed mapping to describe relationships between physical habitat and associated biota include a study using aerial photography to define the Wairarapa coast's vulnerability to climate and other anthropogenic changes (Robertson & Stevens, 2007). Acoustic and video surveys identified large-scale biotope turnover along New Zealand's coastline (Compton et al., 2012), and reserve-related habitat changes in northeastern New Zealand (Leleu et al., 2012). The relationship between Taranaki's kelp *Ecklonia radiata* and physical variables was identified by a drop-camera survey (Crofskey, 2007).

2. Study area description: the coastal environment and urban influences

Wellington's south coast (WSC) lies ~ 8km south of the city centre and extends west 11.4km linearly (coastline contour ~19.3km) from Point Dorset at the

harbour entrance to the Sinclair Head area (Figure 1.1). The built coastal environment includes residential housing, light industry and urban infrastructure extending west to an old quarry, decommissioned in 2000 (black arrow, Figure 1.1). West of this area is undeveloped scrubland with fewer than 10 holiday homes and ephemeral or low-flow streams. At the eastern end of the south coast, streams have been built into the storm water system. Within this area lies the 855-hectare Taputeranga Marine Reserve, located between the old quarry to the west and Princess Bay to the east and extending south of the mean high water spring mark to a depth of ~40m.

WSC is an exposed shoreline composed of a series of sedimentary greywacke cliff headlands separated by sand-gravel embayments (Figure 1.2) (Morton, 2004; Grapes & Campbell, 2008). Seabed structure and composition are submarine extensions of this alternating hard/soft shoreline geomorphology. Headlands extend to submarine hard-bottom mosaics composed of topographically complex platforms and outcrops with gullies, overhangs, holes, and channels. The seabed is structured by southerly oceanic swells, storm waves, surges, large-scale currents, hydraulic pressure, and sediment scour inshore (Carter & Lewis, 1995; Carter, 2008). Rocky reef areas are heterogeneous, patchy, and separated by aprons of mobile substrata that include boulder, cobble, gravel, pebble and sand. At shallower depths, macro-algal species can shift mobile sediments and erode bedrock (Scheibling et al., 2009; Garden & Smith, 2011). Finer sediments are exported by strong tides. Submarine platforms have small slopes and extend up to 2.5km offshore to depths of ~50+m and bathymetry follows an east-west axis. (Morton, 2004; Gardner, et al., 2008; Grapes & Goh, 2008).

Local topography funnels wind along a north/south axis through the Cook Strait to create a dynamic wind climate that affects subtidal systems shallower than 50m: winds greater than 15 knots average 124 days/year and gale-force southerlies average 150 hours/year. The mean maximum tide height is low at 1.7m. Bottom currents range in speed from a residual westward rate of ~4.0cm/sec to inshore rates of ~50cm/sec with a maximum observed rate of ~100cm/sec. Mobile seabed substrata are moved on timescales ranging from short (minutes) to decadal by storms, southerly swells, and tides (Gardner, et al., 2008; Grapes & Goh, 2008).

The area lies in the North Cook Strait coastal biogeographic province, an ecotone between warmer northern and cooler southern waters. Five oceanic currents contribute to this species-distribution boundary (Apte & Gardner, 2002; Waters & Roy, 2004; Shears, et al., 2008). The Westland brings warmer Tasman Sea water north along the west coast where it is carried through Cook Strait by the

D'Urville current. The Southland Current carries warm Tasman and cold sub-Antarctic water north along the east coast while the East Cape Current carries warmer East Australian Current water south and east of the Canterbury Current. This mix is retroflected east of the Strait but can be drawn into the eastern Strait's entrance (Figure 1.1, left inset) (Morton, 2004; Grapes & Campbell, 2008; Grapes & Goh, 2008).

Sources of contaminants along the south shore include run-off from three open landfills and an urban stream near Owhiro Bay, run-off from a closed landfill at Houghton Bay, and the storm water infrastructure along the eastern shore. Storm-related overflows typically occur at Owhiro, Lyall and Island Bays. The Moa Point Wastewater Treatment Plant outfall lies 1.8km off Moa Point but a recent survey of water quality and soft-bottom assemblage structure detected no noticeable impact on the local benthic community (Cameron, 2007).

Water quality along the south coast and in Wellington Harbour is monitored during the summer recreational beach season by the Greater Wellington Regional Council. South coast sites (8 in total) are monitored for the indicator group *Enterocci* and, in Wellington Harbour, for other contaminants (Milne, 2010). Coastal water quality compliance with national recreational beach standards was high (80-100% compliance) during the last reported sampling period (2005-10). Contamination is often linked to rainfall or high winds within the 24 hours preceding the event but typically disperses within 48 hours of detection (Greenfield et al., 2012; Oliver & Milne, 2012). Two natural sources of *Enterococci* along the south coast have been suggested: the large resident seabird population at Owhiro Bay and decaying seaweed, although the link between this and *Enterocci* was not explained (Oliver & Milne, 2012). The rate at which sediments are transported out of the harbour and along the south coast by long-shore currents has not been quantified. That a south coast site (Breaker Bay) lying directly outside the harbour entrance did not experience any contamination events during the above five-year report period suggests that any discharged harbour pollutants have a negligible effect on south coast nearshore assemblages.

Figure 1.1. Wellington south coast overview. *Inset, left:* oceanic currents contributing to the unique local benthic ecology. *Inset, right:* overview of the region's geography. The blue polygon represents the study area and the green polygon marks the boundaries of Taputeranga Marine Reserve, gazetted in September 2009.

Figure 1.2. Urban development along Wellington's south coast, as indicated by the extent of the sewer/storm water infrastructure (purple, yellow lines). Black labels (main photo) and white labels (inset) identify major embayments referred to in this study. Blue labels identify main headlands. *Inset:* Detailed view of Breaker Bay sites assessed in this study. Sewer and stormwater data owned by Wellington City Council.

3. Thesis structure

3.1 Thesis aims

Marine nearshore rocky-reef zones are characterized by dynamic, high-diversity, structurally complex communities at the interface of terrestrial and oceanic processes. Effective management and conservation of resources in this zone require knowledge of biophysical patterns in these subtidal assemblages, and of connections between terrestrial and marine processes. The process of acquiring this information in underwater environments is time-consuming and costly, necessitating the use of survey methods that can assess ecosystem structure and status and identify possible impacts with a minimum of time, effort, equipment, crew, risk, and cost. Methods must also be repeatable by different surveying groups. Landscape ecology provides a conceptual framework for the study of coastal systems through an emphasis on spatially-referenced data and scale in exploring biological patterns and processes (such as connectivity and migration) and associated physico-chemical processes. Such a spatially-explicit approach allows a more nuanced interpretation of biophysical patterns and interactions than afforded by traditional statistical approaches in which spatial relationships are modelled as discrete factors (Anselin *et al.*, 2006; Pittman *et al.*, 2011).

The goal of this investigation was to investigate how WSC's shallow rocky-reef epibenthic communities and diversity vary spatially and temporally, which species/groups determine these patterns, what and how much physical habitat is available and how it contributes to community variation, and how environmental variables contribute to community structure. As well as contributing to an understanding of how temperate rocky-reef epibenthic habitats function, and the scale at which such patterns can occur, this work also provides to resource management agencies information on aspects of nearshore benthic ecology ranging from benchmark data on biota and the contributions of natural and anthropogenic variables; knowledge of representative species; diversity patterns; descriptions of physical habitat type, distribution, abundance, and biotic associations in the form of benthic habitat maps developed in a geographical information systems (GIS) framework; and tools for ecosystem modelling.

This goal was addressed using direct and remote survey techniques. Community-based multivariate analyses were used to identify the species/taxa driving spatial differences in assemblages, representative subsets to streamline future monitoring efforts, and spatial patterns in biodiversity. Benthic habitat mapping tested the effects of sampling intensity and visual analysis methods on the detection of available habitat and abundance of associated benthic cover. Tests of

association and geostatistical methods assessed preferences and spatial trends in benthic canopy cover.

The patterns and the long-term trends identified here, in relation to two previous pre-reserve baseline studies (Cole & Jackson 1989; Pande, 2000) targeting only a few species, will serve as essential benchmark data for the assessment of local reserve effects, the rate at which local impacted species recover from long-term harvesting pressure, and the effect of this process on local ecosystem structure. Results can also be used to develop and test hypotheses on processes structuring local assemblages, used in predictive modelling of anthropogenic and natural changes to species and communities of interest, and can inform adaptive management strategies for WSC.

3.2 Chapter aims

- Chapter 2 examines patterns in physical processes and physical habitat by looking at spatial and interannual variability in WSC environmental variables, including shallow substratum composition and distribution.
- Chapter 3 examines temperate rocky-reef epibenthic community dynamics using two annual baseline surveys to describe abundance, composition, representativeness, and size-frequency patterns in WSC nearshore epibenthic rocky-reef assemblages. Taxa responsible for spatial patterns and subsets of representative species for long-term monitoring are identified. The contribution of environmental factors and coastal processes to assemblage structure is evaluated.
- Chapter 4 describes the scale at which changes in WSC nearshore rocky-reef diversity occur, and assesses turnover in taxonomic distinctness. These are assessed in relation to the local environmental characteristics identified in chapter 2.
- Chapter 5 uses patterns in size-frequency as a proxy for productivity patterns and for the possible influence of environmental factors on growth in key macro-algal and mobile macro-invertebrate species. These data are then modelled to develop predictive regression equations for use in future non-destructive ecosystem modelling.
- Chapter 6 uses a drop camera to identify nearshore benthic resources and seafloor structure to develop benthic habitat maps at different thematic resolution in a GIS framework. It contributes to the field of remote surveying by assessing the impact of sampling intensity and visual image

analysis methods (categorical vs semi-quantitative) on the identification of benthic resources.

- Chapter 7 reviews these results in terms of past research and future directions.

Chapter 2. Variability in Wellington south coast's nearshore rocky reef environment

Fujin, Japanese god of wind

1. Introduction

Evaluating coastal marine ecosystems for resource management and conservation is challenging due to the dynamic, complex, hierarchical, and interrelated relationships of the biotic and abiotic components (Tian et al. 2011). However, long-term monitoring of environmental conditions in coastal areas is considered an essential prerequisite to effective management practices because of increasing human impacts on coastal ecosystems worldwide, and because of the links between environmental parameters and biological status of coastal marine ecosystems. Knowing the levels of environmental parameters that support desired levels of biological goods and services allows resource managers to implement strategies that can maintain and support these parameters (Salomidi et al. 2012; Rombouts et al. 2013). Traditional ecosystem measures include abundance, diversity, distribution, size, and taxonomy, but the ecosystem-based management approaches currently in widespread use require models that synthesize environmental and biological processes and patterns in order to develop effective management strategies (Rees et al. 2008; Ferreira et al. 2011; Tian et al. 2011).

Selecting parameters to use as monitoring and assessment measures is challenging. Currently over 200 ecological indicators have been used in assessment studies (Borja and Dauer, 2008; Rombouts et al. 2013). Selection is a function of the question(s) being asked and the scale at which they are defined, with questions concerning individuals, species, or populations typically requiring different ecological indicators than those addressing an entire community or ecosystem (Borja and Dauer, 2008). Moreover, a multivariate approach to sampling will account for the complementarity of many environmental variables (Rakocinski, 2012). Qualities of useful and appropriate indicators are sensitivity to change at the required thresholds; responsiveness to the stressor; occurrence throughout the area of interest; ecological relevance; ease of measurement; and simplicity and affordability. They will also allow discrimination between naturally-occurring patterns and those driven by anthropogenic activity (Rees et al. 2008; Rombouts et al. 2013).

Anthropogenic environmental factors identified as having negative effects on benthic macro-algae and macro-invertebrates in studies on temperate nearshore communities in northern and southern hemisphere systems include ocean warming; ocean acidification; sea-level rise; habitat degradation and destruction from agricultural, commercial, industrial, and urban activity; eutrophication; seabed mineral extraction; and industrial, agricultural, thermal and light pollution (Olsgard et al. 2008; Connell and Russell, 2010; Subida et al. 2012; Tamburello et al. 2012; Suikanen et al. 2013). In addition, natural structuring processes include storms, tides, currents, fetch, wave energy and surge, seafloor scour and landslides, and erosion (Heath, 1979, 1986; Cole, 2002; Roberts et al., 2006; Score et al., 2012). Other abiotic factors and processes that occur naturally but whose effects can be amplified by coastal human activity include physico-chemical and geological variables: temperature, salinity, dissolved oxygen and nutrients, pH, light, space, fetch, sedimentation, and substratum type and complexity (Mann, 2000; Burrows and Robb, 2008; Grober-Dunsmore et al. 2008; Tann et al. 2009).

Ecological indicators with the potential to be affected by altered levels in physico-chemical indicators include primary productivity, nutrient cycling, energy flow,

trophic interactions, resistance to disease and invasive species, biodiversity, size, and abundance (Roberts et al., 1998). Survival and respiratory rates of two ascidian species showed significant responses to experimental changes in salinity, dissolved oxygen, and temperature (Lenz et al., 2011), while Australian rocky reef sponge function responded to altered salinity, light, and sediment (Roberts et al., 2006). Sewage outfall nutrients were linked with altered macro-algal community structure in a shallow Mediterranean rocky-reef community (Arevalo et al., 2007) and detected in a New Zealand rocky-reef food chain (Dudley & Shima, 2010). Seafloor composition, complexity, and orientation have been correlated with increased survivorship and abundance of fish, polychaetes, and barnacles in experimental and field experiments, as well as with sponge assemblage structure (Wahl & Hoppe, 2002; Gratwicke & Speight, 2005a, 2005b; Scharf et al., 2006; Càrdenas et al., 2012).

The indicators selected for monitoring can be few and basic yet provide essential information: temperature, dissolved oxygen, salinity, water clarity, and seabed characteristics can provide information on community status as well as information on the environmental thresholds of key species. Temperature, dissolved oxygen, and salinity are among the primary physico-chemical factors regulating species' occurrence, their metabolic functions, life cycles, and resistance to disease (Rosenzweig, 1995; Ray, 1996). Thermally stressed algal and invertebrate communities can undergo bleaching and subsequent phase shifts (Vilchis et al. 2005; Campbell et al., 2011; Wernberg et al. 2011), while hypoxic conditions can signal eutrophication, also a factor in community phase shifts (Rakocinski, 2012). Hyper- and hyposalinity can indicate altered coastal flow regimes due to coastal alterations (Trabelsi et al. 2013). Reduced water clarity can indicate increased anthropogenic sedimentation and can limit light levels, which in turn affects macro-algal productivity (Williams et al.; Suikanen et al. 2013; Trabelsi et al. 2013).

1.1 Local studies on Wellington south coast's environmental variables

Previous research on Wellington south coast's (WSC) nearshore environment has addressed two contributions to nearshore physico-chemical patterns: human impacts on water quality through coastal urban development, and naturally occurring dynamics. Monitoring of water quality at local recreational beaches along the south coast using microbiological indices began about twenty years ago, with additional indices later added (sediment and shellfish tissue levels) (Milne & Wyatt, 2006). Annually, at least one instance of contamination has been identified at one or more south coast beaches during regional council monitoring, typically at Owhiro Bay, Island Bay, or Lyall Bay (Milne, 2009; Ryan & Warr, 2008, 2010). In Wellington Harbour, stormwater run-off was also identified as a pollution source that included heavy metals, polycyclic aromatic hydrocarbons, and DDT (Milne, 2010). These council studies were limited to testing for pollutant presence so did not study the effects of point-source pollution on nearshore marine assemblages (Milne & Watts, 2008). Natural environmental processes identified in council reports as contributing to nearshore water quality included rainfall, wind direction, current, and presence of algal cover. Rainfall typically preceded microbial contamination of recreational beaches, while foliose macro-algal cover appeared to sequester microbial pollutants (Milne, 2009). The south coast beaches most often affected by exceedance events

were in highly developed coastal areas with several stormwater outfalls (Ryan & Warr, 2010).

Local studies on naturally occurring interactions between the physical and biotic components of Wellington south coast's (WSC) nearshore epibenthic rocky-reef assemblages include the development of biogeographic regions of New Zealand's shallow rocky reefs using environmental variables. Water clarity, a measure of turbidity or suspended sediment, and fetch were the two environmental variables most strongly identified with rocky-reef community structure locally (Shears & Babcock, 2007). Cook Strait hydrology was identified as a probable block to gene flow between northern and southern populations of two local species, the starfish *Patiriella regularis* and the green mussel *Perna canaliculus* (Apte & Gardner, 2002; Waters & Roy, 2004). Along WSC, sponge community structure was correlated with substratum inclination (Càrdenas et al. 2012).

1.2 Chapter aims

My aims in this chapter were to 1) identify spatial and seasonal patterns in a subset of environmental parameters vulnerable to the impacts of climate change (IPCC, 2013) and 2) assess the homogeneity of the nearshore shallow rocky-reef physical habitat (seabed). Environmental data would then be used in the next chapter to identify environmental contributions to biological communities. This chapter provided updated information on patterns in physico-chemical variables along a greater spatial scale than addressed by previous studies, and 2) quantified for the first time the composition, range, and distribution of physical habitat (seabed classes) along Wellington south coast. Specifically, I asked if a shallow (<10m) inshore thermocline, noticed by this study's divers on shore dives, extended to the depths (5-13.6m) of the nearshore epibenthic community surveyed in Chapter 3. This was accomplished by testing for differences in water column position (seabed versus surface) in three key variables: dissolved oxygen, salinity, and temperature. Secondly, I tested for temporal (seasonal and annual) and spatial (inter-site) differences in those variables and in water clarity. Water clarity was assessed vertically (Secchi depth) and along the seabed (horizontal Secchi distance). Thirdly, I assessed the homogeneity of the physical habitat "reef" by testing for differences in hard-bottom composition, distribution, and seabed profile (rugosity) of the south coast's shallow rocky reefs. This information will provide data that can be used firstly to model and test hypotheses about the link between local environmental variables and biotic status. Results could then be used to model impact assessment scenarios as part of long-term management plan. These results also provide a benchmark of basic water quality parameters for long-term monitoring of for natural and anthropogenic changes.

2. Methods

2.1 Field methods

Temperature, salinity, dissolved oxygen, and water clarity (Secchi depth) readings were measured from a boat at monthly intervals between October 2007 and October 2009 at eight sites off WSC (see Chapter 3 survey sites description; Figure 3.1). Readings were taken at two levels in the water column: ~1m below the surface and at

~1-2m above the seabed depending on sea conditions. The depths of readings were affected by weather and sea conditions at each site, with a mean depth of 9.2m (3.5-17.1). Wind/sea conditions prevented sampling on three occasions (n=21) and salinity readings were not collected on an additional two occasions (n=19). Eight of the ten epibenthic community survey sites described in Chapter 3 were surveyed (Figure 2.1), with the westernmost site (2nd Wash) beyond the boat's operational limit. At each site, measurements were taken at three haphazardly selected locations to ensure consistent meter readings and these readings were later averaged for analyses. Trips were made between 0800 and 1600, beginning on a rising tide at the westernmost site and finishing at the easternmost site at the turn of the tide. Exceptions included three trips that started at sunrise and four that sampled the easternmost site (Barrett Reef) in the first third of a falling tide. Trips were started when northerly/southerly winds were less than 25 knots and seas 2m or less.

Salinity was measured with a YSI 30 conductivity-salinity-temperature meter (precision $\pm 1\%$ practical salinity units (psu) / $\pm 0.1^\circ\text{C}$). Dissolved oxygen and temperature were measured with an YSI 550 meter (precision ± 0.3 mg/L/ $\pm 0.2^\circ\text{C}$). Depth was obtained with a Norcross 2200PX hand-held depth sounder ($\pm 10\text{cm}$ precision). Temperatures from the two meters were cross checked to ensure proper functioning of the units. Water clarity (turbidity) was assessed with a 24-cm Secchi disc (Davies-Colley et al. 1993) and values converted to ratios by dividing Secchi depth by absolute depth. As a measure of seabed turbidity, horizontal Secchi distances were measured during two annual dived epibenthic baseline surveys (summer, 2007/08 and 2008/09) to complement the vertical Secchi profiles of vertical water column clarity. These values were obtained by averaging the distance at which a Secchi disc held vertically at one end of a transect tape disappeared, then reappeared to a diver swimming along the tape (Steel & Neuhauser, unpublished).

Rugosity was measured at the ten baseline survey sites described in Chapter 3. Survey depth averaged 8.0 m (range 4-13.2m). At each site, 3 locations were haphazardly selected as hubs for three 20-metre survey transects per hub (n=9/site). To avoid overlapping transects and ensure independence among hubs/transects, distances among hubs were measured remotely or with a Garmin 72 GPS unit and each transect was started at the 5m mark to increase separation among transect lines. Sampling areas covered 1-2 hectares per site, with the baseline survey markers in Chapter 3's Figure 3.1 indicating the approximate starting points of the rugosity transects. At each transect a 20m linear distance was measured with a tightly laid survey tape. Along this transect a weighted lined was then draped over seabed contours to measure seabed profile (Figure 2.3A). Transect headings were randomly generated with different numbers used at each site. Headings less than 10° apart were rounded up to avoid overlapping transects. Dividing the surface distance by the linear distance yielded the rugosity ratio (Kingsford & Battershill, 2003).

Substratum composition (physical habitat), defined by seabed class, was classified at the ten sites used for the two annual epibenthic surveys along Wellington's south coast that are the focus of Chapter 3. Substrata categories were logged every metre at three points widthways along each of four 25-metre x 2 metre transects at each site (n=300; Figure 2.3B). Headings for each transect were randomly generated. Classifications used the Wentworth sediment scheme (Wentworth, 1922) and the bedrock class "Reef", which included "Patch reef", arbitrarily defined as

bedrock less than ~5m diameter. During the second annual epibenthic survey, reef orientation (horizontal, vertical, inclined) was also noted (n=300/site) (Kingsford & Battershill, 2003).

2.2 Statistical analyses

PASW Statistics v18 (SPSS) was used for analyses. Data were first assessed for parametric test assumptions (linearity, normality and homoscedasticity). P-values for all tests were set at $\alpha = 0.05$. The effect of position in the water column (surface vs. seabed) on temperature, salinity, and dissolved oxygen at the 8 monthly sites was tested with Mann-Whitney U tests because data did not meet parametric test assumptions (Quinn & Keough, 2006). Spatial and temporal differences in temperature, salinity, dissolved oxygen, vertical Secchi depth and horizontal Secchi distance among sites (8-10), seasons (4), and year (2007-08, 2008-09) were tested with generalised linear models (GzLMs) because this family of tests accommodates factorial designs for non-normal, non-linear distributions (Norušis, 2010). A normal distribution with identity link function was used to test for differences among main effects (site, season, year) and all interactions (Bolker et al., 2003; Quinn & Keough, 2006). Horizontal Secchi distance data were pooled across years to compensate for irregular sampling across sites, seasons, and years. Therefore, the only main effects tested for this data set were "Site" and "Season". Differences in rugosity among sites (10) were tested using a GLM (Norušis, 2010).

To identify associations between sites and seabed composition, Pearson χ^2 tests were used to test two associations (Gotelli & Ellison, 2004). The first association tested was between site and seabed composition for data collected along randomly oriented transects at ten sites during two annual epibenthic surveys (2007/08 and 2008/09; see Chapter 3 for discussion of surveys). Seabed classes were classified according to Wentworth's sediment classes (Wentworth, 1922), then pooled into classes comprising a minimum of 5% of the total material at each site. The final five seabed classes thus derived were: reef, reef/boulder in rubble, boulders in cobble/sand, cobble in pebble/sand, and sand/gravel. The second test of association was between site and the seabed class "reef" (bedrock) orientation: horizontal, inclined, vertical.

To identify which classes contributed to differences among sites, Pearson residuals were used, with higher/lower values indicating stronger/weaker associations among classes and site (Gotelli & Ellison, 2004). Tests used randomly collected observations of seabed class pooled across both survey years. A separate analysis for association between "Site" and "Reef" orientation was performed because this characteristic was logged only during the second year's survey.

To identify a significant association between turbidity in the water column and in the bottom layer, a Pearson correlation test was conducted between Secchi depth and horizontal Secchi distance (Gotelli & Ellison, 2004). Because these two variables were sampled on different dates, a subset of the Secchi depth data was used if their collection dates coresponded to horizontal Secchi distance collection dates ± 1 week. Horizontal and vertical Secchi data were also matched by site (n=32).

3. Results

3.1 Aim 1: Vertical gradients in temperature, salinity, and dissolved oxygen

No differences between in nearshore shallow water column position (sea surface vs. seabed) were detected in temperature, dissolved oxygen, or salinity in the depth range surveyed ($\bar{x}$ = 9.2m, range 3.5-17.1m) at the eight sites surveyed along Wellington's south coast ($p \geq 0.392$; Table 4.1). Median values (minimum, maximum) for variables across both sea strata were: temperature = 13.9° (10.0-18.6); dissolved oxygen = 8.01 mg/L (6.4-11.8); and salinity 34.4 psu (31.1-35.5). Consequently, water surface values alone for these three variables were used in further analyses.

3.2 Aim 2: Spatio-temporal trends in water parameters

3.2.1 Temperature, dissolved oxygen, and salinity

Temporal patterns in sea temperature were identified. Sea surface temperature differed significantly among the four seasons (see Table 2.2 for test statistics), with the highest values observed in summer (15.6°C) and lowest in winter (11.6°C). Temperature also differed significantly between the two years, with a mean of 14.2° in 2007/08 and 12.9° in 2008/09. A significant interaction between "Season" and "Year" was detected (Figure 2.4) but the interaction was not strong: temperatures followed the same pattern of highest values in summer and lowest in winter in both years. The interaction was driven by higher temperatures in 2007/08 for spring and autumn than in 2008/09. No differences among sites were identified.

Temporal differences in dissolved oxygen were identified, with significant differences identified for the main effects "Season" and "Year" (Table 2.2). Highest values occurred in spring and winter (8.7 mg/L for both seasons) and lowest in autumn and summer (7.8 mg/L and 7.9 mg/L respectively), with an overall range of 4.4 mg/L. Mean values differed by less than 1 mg/L in 2007-08 (8.4 mg/L) than in the following year (8.1mg/L). "Year" by "Season" differed significantly (Table 2.2, Figure 2.4) with only autumn values similar. Values for the other three seasons varied by year but the maximum range was only 1.5 mg/L. No significant differences among sites were identified ($p > 0.5$, Table 2.2).

Salinity values differed significantly among sites, seasons, and years (see Table 2.2 for test statistics). The greatest difference in salinity among sites was less than 1 unit, at 0.3 psu (1st Wash/Yung Pen highest, Barrett Reef lowest). The range in seasonal values was equally small at 0.5 psu, with the highest mean in autumn (34.6 psu) and lowest in spring (34.2 psu). A small difference of 0.5 psu was detected between the two years. "Site" did not vary with "Year" but did with "Season" (Figure 2.4). This interaction was driven by a lower salinity in the first year's winter. The interaction "Season" x "Site" was characterised by steady levels (less than 1 psu difference) across most sites, with a slight drop at the eastern end (Barrett Reef) for spring, summer, and autumn and a sharp drop (32.8 psu) in winter. The overall range in salinity values for this interaction was only 1.5 psu. A significant third-order interaction was also identified, but the biological significance of higher order interactions is notoriously difficult to decipher so those results are not addressed here (Tabachnik & Fidell, 2007).

3.2.2 Variability in water clarity: Secchi depth and horizontal Secchi distance

Secchi depth also differed significantly among sites, seasons, and years (see Table 2.2 for test statistics; Figure 2.5). Values showed no clear spatial pattern among sites, fluctuating from 52.3% at the westernmost site (1st Wash) to a high of 59.8% at Palmer Head, with the lowest observed value, 39.5%, at the harbour entrance (Barrett Reef). The "Season" factor identified a high Secchi depth of 60.2% in summer and a low of 39.7% in winter. The "Year" factor identified lower values in the first year, 49.8%, than in the second year (57.2%). Only one interaction, "Season x Year", was significant (Figure 2.5). Here the seasonal value in the first year was lowest during summer before rising to similar values for autumn and winter, while in the second year Secchi depth was greatest in summer before dropping in autumn and again in winter. No spatial interaction was detected. The overall mean Secchi depth was 54.1% and highly variable, with a range from 6 to 100%.

Horizontal Secchi distance, measured along the seabed, differed significantly among sites and among seasons (see Table 2.2 for test statistics; Figure 2.5). Values mirrored the vertical Secchi depth patterns measured at the western and central site, with a mid-range value of 6.8m at the westernmost site (1st Wash), rising to a high of 8.8m at Yung Pen, then falling to a low of 4.0m at Flax Bay. A slightly higher value of 4.8m was measured at the harbour entrance (Barrett Reef). Seasonally, spring and summer values were mid-range (6.2m and 6.8m), with a high of 7.7m in autumn and a low of 4.3m in winter. The overall mean horizontal Secchi distance was 3.4m (1.0-15.0). No interaction between the two factors was detected.

Water column turbidity (Secchi depth) and seabed turbidity (horizontal Secchi distance) did not appear to be related ($r = 0.1$, $p = 0.49$, $n = 32$) for the dates and sites tested.

3.3 Aim 3: Spatial differences in physical habitat along Wellington's south coast: Seabed profile and composition

These results showed that WSC rocky reefs are not a homogenous expanse of unbroken bedrock but rather a heterogeneous and structurally complex environment in which mobile sediments abut channelled bedrock of variable relief and orientation.

3.3.1 Seabed profile/rugosity

Seabed profile differed significantly among sites (see Table 2.2 for test statistics). The central site "The Sirens" was the most rugose, with an estimated mean of 2.3, while the flattest site was observed at the eastern Flax Bay site, with an estimated mean of 1.1. Means for the remaining eight sites fell into a narrow range between 1.5 (Yung Pen) to 1.6 (Red Rocks) (Figure 2.5), highlighting the similarity of seabed profiles along the majority of the nearshore area surveyed. The overall mean rugosity ratio was 1.5 (1.1-2.9).

3.3.2. Spatial patterns in seabed composition

Strong associations between sites and seabed composition were identified (see Table 2.2 for test statistics), indicating that seabed composition varied significantly among sites. "Reef" comprised over 51 % of habitat in all but one of the sites, with the eastern sites Breaker Bay highest at 93% and Flax Bay lowest with only 13% reef. "Reef/boulder in rubble" also had highly variable distribution, ranging from none observed at six sites to 44% at the central site of Princess Bay. Distribution of "boulders in cobble/sand" ranged from 7% at two sites (the western 2nd Wash and the eastern Breaker Bay) to 79% at the eastern Flax Bay. "Cobble in pebble/sand" comprised no more than 20% of central The Sirens site and no more than 1% of eight other sites. "Sand" comprised no more than 7% of habitat surveyed in the rocky reef zones (Figure 4.6). Sites characterised by the highest residual values (Table 2.3A) included The Sirens (central), with "cobble in pebble/sand" (19.1), Princess Bay (central) with "reef/boulder in rubble" (9.9), Flax Bay (east) with "boulders in cobble/sand" (26.2), and 2nd Wash (west) with "reef" = 6.3 and negatively associated with "boulders in cobble/sand" (-8.9).

Reef orientation was predominantly horizontal, with the lowest proportion observed at the central site Princess Bay (48%). The highest proportion was observed at the eastern site Yung Pen (100%), followed by the western site 1st Wash (94%) and the eastern site Barrett Reef (76%). The highest proportions of vertically oriented reef occurred at the central sites Princess Bay (58%) and The Sirens (44%). Lowest proportions were found at Flax Bay (0%), the western sites 1st Wash (6%) and 2nd Wash (95%)(Figure 2.6, Table 2.3A, B).

4. Discussion

The goals of this study were to quantify patterns in the environmental component of WSC nearshore rocky-reef communities, and secondly to characterise physical habitat through an assessment of seabed composition and profile. It is the first study to survey a set of physico-chemical variables along the entire length of Wellington's south coast and the first to assess the homogeneity of the area's reef structures. It also provides updated baseline information on patterns in temperature, salinity, dissolved oxygen, and water clarity. The parameter levels identified here did not reach values identified elsewhere as having deleterious effects on the metabolism, growth, distribution, survival, or abundance of temperate benthic macro-algae, mobile or sessile macro-invertebrates (Gall et al., 2012; Karsten et al., 1996; Roberts et al., 1998; Smith et al., 2009; Wahl & Hoppe, 2002; Witman & Grange, 1998).

The lack of significant differences between seabed and surface values for temperature, salinity, and dissolved oxygen in shallow nearshore water parcels is to offshore Cook Strait homogeneity, which is well-mixed by semi-diurnal tides, wind, currents, and storms (Heath, 1970, 1979, 1986). This result suggests that the thermoclines encountered during shore dives by this study's divers over a two-year period (2007-09) at five baseline survey sites (Yung Pen, The Sirens, Princess Bay, Flax Bay, Breaker Bay) were highly localised and ephemeral.

Trends observed in the main effects of "Season" and "Year" for sea temperature, dissolved oxygen, and salinity reflect natural seasonal processes. High summer and low winter sea surface temperatures reflect atmospheric temperatures,

while low summer and high winter dissolved oxygen levels are a function of dissolved gas/liquid dynamics (Score et al., 2012). Stable salinity levels with only minimal variation in value indicate minimal impacts from land run-off (Lenz et al., 2011). The slightly lower values (< 1 psu) noted at the harbour entrance site (Barrett Reef) in the "Season x Site" plot suggest the presence of a small fresh-water surface effect that was probably due to rainfall preceding and during monitoring trips. The lowest observed salinity value of 31.2 psu was measured directly after a heavy rainfall and this relationship between surface salinity values and rainfall was noted by Milne (2009). Observed salinity values correspond to ranges supporting high coastal biodiversity (Devlin et al., 2007; Karsten et al., 1996; Zettler et al., 2007). Given the very narrow range of salinity values observed, significances here are probably statistical rather than practical.

The lowest dissolved oxygen value of 6.6 mg/L was well above the boundary of 5 mg/L identified by the European Union Water Directive Framework as indicative of poor water quality (Devlin et al., 2007). The narrow range in the three variables suggest that human impacts from the Hutt River at the north end of Wellington Harbour, harbour run-off and stormwater outfalls have limited effects on south shore surface layers. Water column mixing and water movement due to Cook Strait hydrology appear to homogenize harbour water flow and discharge (Carter, 2008). Interannual stability of dissolved oxygen levels is suggested by the similarity of values observed at a central south coast site during a 1998-2000 study (Helson et al., 2007) and these values.

Spatial variation in water clarity was driven by consistently lower values at the harbour entrance (Barrett Reef). Natural factors affecting turbidity include seasonally-fluctuating zoo- and phytoplankton levels (Sanden & Hakansson, 1996; Valiela, 1995). Possible anthropogenic contributors to higher turbidity at this site include road and storm-water run-off from the urbanised inner-harbour area, sediment carried by Hutt River discharge, and sand and sediment transported by two rivers east of the harbour entrance, the Orongorongo and Wainuiomata Rivers (Carter, 2008; Milne & Wyatt, 2006). Secchi depths indicated that these effects were limited to the east end of Wellington's south coast. The "Season x Year" interaction suggests that the above processes were characterised by interannual variation due perhaps in part to natural hydro-meteorological fluctuations (Comín et al., 2004), if not also variability in the anthropogenic factors. An additional possible contributor to spatial patterns in turbidity is the process of sediment resuspension, due to site-specific interactions between waves and currents (Carter, 2008).

The lack of correlation between water clarity (turbidity) along the seabed (horizontal Secchi distance) and in the water column (Secchi depth) suggests an uncoupling of the local-scale hydrological processes that drive suspended sediment transport. This might be due to differences between the area's surface winds, tides, currents and seabed currents (Carter, 2008; Heath, 1986). Such local-scale differences in turbidity due to currents and eddies have the potential to affect benthic community structure; suspended sediments contributed to mortality in macro-invertebrate larvae of *Haliotis iris* and *Evechinus chloroticus* (Denny, 1987; Phillips and Shima, 2006). These patterns should be interpreted with caution, given the differences in collection dates for the two data sets because WSC conditions change with tides, sudden weather shifts, and atmospheric influences (Carter, 2008).

WSC rocky reefs are not homogeneous habitats of bedrock platforms. They were identified as heterogeneous arrangements of mobile sediment and bedrock consisting of platforms and reef patches of primarily horizontal orientation abutting fields of variously-sized mobile substrata. Significant differences in rugosity (seabed profile), seabed composition, and reef orientation defined sites, but a linear spatial gradient was not identified. The highest rugosity value occurred at the central Sirens site and the lowest at the eastern Flax Bay site. The most variable component of the south coast rocky-reef substratum was the proportion of mobile classes (boulders and smaller) at each site. Physical habitat (seabed) composition, orientation, micro-topography, geochemistry, and availability influence benthic community structure and processes through these substratum characteristics and through the influence of seabed profile on local hydrology (Perkel-Finkel and Airoidi, 2010). Outcomes of habitat loss include phase shifts, reduced productivity, loss of ecosystem goods and services, and lower benthic biodiversity (Airoidi et al., 2008; Anderson & Millar, 2004; Bulleri & Benedetti-Cecchi, 2006; Connell & Irving, 2008; Dobson et al., 2006; Hewitt et al., 2005; Lapointe & Bourget, 1999).

Environmental and ecological sampling strategies are often limited by lack of funds, field and lab testing equipment, and investigator inexperience. Often only basic monitoring equipment is available to researchers unfamiliar with the study area or its environmental dynamics (Birk et al., 2012). This study was limited to the evaluation of basic water parameters and by the investigator's lack of experience. Such a small-scale, intensive sampling effort was not necessary to capture spatial and seasonal trends, and future sampling can focus time, effort, and funds on sampling WSC water quality at one site. The limited set of water parameters studied were, however, valid choices for assessing water quality because spatio-temporal patterns in surface temperature, dissolved oxygen, salinity, and water clarity have all served as valid and critical indicators of negative changes in temperate and tropical benthic ecosystem function and structure (Rombouts et al. 2013). Future sampling along WSC might also benefit from including environmental parameters identified in other monitoring studies as reliable indicators of ecosystem function, eutrophication, habitat degradation, and pollution, such as chlorophyll a, dissolved inorganic nitrogen and phosphorus, heavy metals, and *Enterocci* (Borja and Dauer, 2008; Bedini and Piazzzi, 2012).

Local field conditions can contribute to confounding factors; in this instance confounding factors were the wide range in depths and distance offshore. High winds and strong surface currents contributed to the wide range in depth, while distance offshore was dictated by the location of the sites, which been determined by a previous baseline survey (Pande, 2000). However, the depth range here matched closely that of the current dived survey because this work was designed to sample conditions at the dived sites. The one depth outlier here, 17m, occurred once, with the majority of depths corresponding to the dived survey depth range. The lack of statistical differences among sites for these parameters suggests that the depth range was not a confounding factor. The range in distances offshore was the legacy of inheriting a predetermined sampling design. The location of Barrett Reef at the mouth of the harbour increased the likelihood of its status as an outlier due to the sediment plumes at the harbour entrance observed after heavy rains. Additionally, the use of hubs for rugosity transects affected transect independence despite the use of

randomly-generated transect headings. An attempt to compensate was made by haphazardly distributing the hubs across each site's reef habitat.

Environmental variable data sets have been used to develop benthic classification schemes, species-distribution models, habitat range and prediction maps, benthic environmental quality indices, and marine value maps. Indicators used in models have included dissolved oxygen, salinity, temperature, metal and organic contaminants, substratum type, depth, slope, sedimentation and biological metrics (Guisan & Thuiller, 2005; Hale & Heltshe, 2008; Holmes et al., 2008; Inglis et al., 2006; Kallimanis et al. 2007). The data is also essential in identifying impacts of climate change (Score et al., 2012). Locally, the inclusion of environmental indicators produced a habitat suitability index that successfully predicted the presence of two marine bivalves (Inglis et al, 2006). Environmental indicators were also used by MAF Biosecurity New Zealand to develop a national marine value mapping database to assist in the detection of invasive species and the assessment of their potential threat to the native marine environment, although environmental data was missing for some coastal cells (MacDiarmid et al., 2008). These results can provide a useful contribution to a long-term local reference database of environmental information.

This study has identified temporal and spatial stability in salinity, temperature, and dissolved oxygen, with water clarity (turbidity) as the most dynamic parameter. These results provide a benchmark set of environmental values that support the current biological status of WSC's nearshore rocky-reef epibenthos. These data will also be useful in exploring multivariate links between environmental patterns and biological community patterns in WSC's rocky-reef epibenthos.

Figure 2.1 Sampling sites (3.5-17.1 metres depth, mean offshore distance 150 metres) of environmental variables in Wellington south coast’s nearshore zone. Extent of the built environment corresponds to the stormwater and sewer network range shown here. Monthly sampling was over a two-year period (2007-09) as part of two annual pre-marine reserve (green polygon) baseline surveys of rocky-reef epibenthic communities. *Aerial photo sourced from Land Information New Zealand orthophoto r27c. Crown copyright reserved.*

Figure 2.2 Sampling design for environmental variables in Wellington south coast’s nearshore zone (2007-09). Water parameters were sampled monthly. “Level” (water layer) factor did not apply to Secchi depth sampling. Rugosity was measured once. Horizontal Secchi distances measures were made during the 2007-09 baseline surveys (chapter 2) so sampling events do not correspond to the monthly vertical Secchi measures.

Figure 2.3 A) Rugosity: Sampling design at 10 sites along Wellington’s south coast: 3 rugosity transects at 3 hubs/site were surveyed (n=9). **B) Physical habitat:** Sampled at the 10 sites: physical habitat class was logged at 75 points along each of 4 transects/site; n = 300) over two annual surveys (chapter 2).

Table

2.1. Mann-Whitney U tests for differences in temperature, dissolved oxygen, and salinity between sea surface and seabed. No significant differences were detected. Variables were sampled monthly between November 2007 and August 2009 at eight sites along Wellington's south coast and values were pooled across sites. P-value was set at $\alpha = 0.05$.

Variable	U	p-value	N	Median
Temperature	84809.0	0.579	833	13.9°C
Dissolved oxygen	84903.0	0.557	833	8.01 mg/L
Salinity	64182.5	0.392	730	34.4 psu

Table 2.2. Spatial and temporal (seasonal, interannual) differences in temperature, dissolved oxygen, salinity, Secchi depth, Secchi distance, rugosity, and seabed composition. Tests used: ¹Generalised linear model; ² General linear model; ³Chi-square test of independence. Data were collected along Wellington's south coast between November 2007 and August 2009. **Bold text** indicates a significant difference at $p = 0.05$.

Variable	Source of variation	Wald χ^2 , ¹	Degrees of freedom	p-value	n
Temperature (°C)	Site (random)	0.61	7	0.999	10
	Season (fixed)	402.33	3	<0.0005	4
	Year (random)	71.94	1	<0.0005	2
	Site * Season	11.68	21	0.948	
	Site * Year	0.77	7	0.998	
	Season * Year	22.46	3	<0.0005	
	Site * Season * Year	4.10	21	1	
N = 278					
Dissolved oxygen (mg/L)	Site	10.81	7	0.144	10
	Season	126.48	3	<0.0005	4
	Year	16.44	1	<0.0005	2
	Site * Season	8.76	21	0.991	
	Site * Year	1.41	7	0.985	
	Season * Year	57.16	3	<0.0005	
	Site * Season * Year	8.93	21	0.990	
N = 278					
Salinity (psu)	Site	39.77	7	<0.0005	10
	Season	102.10	3	<0.0005	4
	Year	176.25	1	<0.0005	2
	Site * Season	35.79	21	<0.0005	
	Site * Year	3.11	7	0.874	
	Season * Year	40.05	3	<0.0005	
	Site * Season * Year	7.89	19	0.988	
N = 246					
Secchi depth (optical depth/absolute depth)	Site	16.84	7	0.018	10
	Season	29.91	3	<0.0005	4
	Year	6.19	1	0.013	2
	Site * Season	21.05	21	0.456	
	Site * Year	4.29	7	0.746	
	Season * Year	97.25	3	<0.0005	
	Site * Season * Year	25.63	21	0.221	
N = 278					
Horizontal Secchi distance (at seabed)	Site	19.43	8	0.013	9
	Season	13.66	3	0.003	4
	Site * Season	17.32	16	0.365	
N = 95					
Rugosity		F ratio ²			
N = 90	Site	18.48	9	<0.0005	10
		χ^2, ³	df	p-value	n
Seabed composition	Site	2443.4	36	<0.0005	5700
Reef orientation	Site	494.1	18	<0.0005	2255

Table 2.3. A) Pearson residuals for χ^2 tests of association between survey site and seabed composition (by percent) in Wellington south coast's nearshore zone. Data were collected during two annual epibenthic baseline surveys (2007/8 and 2008/9). Sites are listed from west (2nd Wash) to east (Barrett Reef). **B)** Reef orientation data were only collected during the second annual survey, so comprise a subset of the "Reef" data in Table A. Orientation was logged for reefs higher than 1 metre. Highlighted cells indicate the highest and lowest residual values observed for each seabed class.

HABITAT SITE	Reef			Reef/boulder in rubble ¹			Boulders in cobble/sand			Cobble in pebble/sand			Sand/gravel		
	Count	Percent	Residual	Count	Percent	Residual	Count	Percent	Residual	Count	Percent	Residual	Count	Percent	Residual
2nd Wash	529	88	6.3	0	0	-3.3	42	7	-8.9	0	0	-4.9	29	5	5.5
1st Wash	488	81	4.2	19	3	2.4	81	14	-5.7	0	0	-4.9	12	2	0.4
Red Rocks	484	81	4.0	0	0	-3.3	112	19	-3.2	0	0	-4.9	4	1	-2.1
Yung Pen	410	68	0.3	0	0	-3.3	95	16	-4.5	90	15	13.4	5	1	-1.8
The Sirens	388	65	-0.8	1	0	-3.0	90	15	-5.0	118	20	19.1	3	1	-2.4
Princess Bay	381	64	-1.1	44	7	9.9	165	28	1.2	7	1	-3.5	3	1	-2.4
Palmer Head	307	51	-4.8	33	6	6.6	259	43	8.8	0	0	-4.9	1	0	-3.0
Flax Bay	77	13	-16.2	2	0	-2.7	473	79	26.2	8	1	-3.3	40	7	8.9
Breaker Bay ²	280	93	5.5	0	0	-2.4	20	7	-6.4	0	0	-3.5	0	0	-2.3
Barrett Reef	486	81	4.1	6	1	-1.5	96	16	-4.5	6	1	-3.7	6	1	-1.5

¹"Rubble" includes (singly or mixed) cobble, pebble, and sand/gravel. n = 600/site (N=5700) except for ²Breaker Bay: n = 300.

B Reef orientation Site	Horizontal			Inclined			Vertical			N
	Count	Percent	Residual	Count	Percent	Residual	Count	Percent	Residual	
2nd Wash	164	59	-0.6	87	32	8.4	25	9	-5.2	276
1st Wash	188	94	5.7	0	0	-5.1	12	6	-5.3	200
Red Rocks	179	69	1.4	21	8	-2.2	58	22	-0.6	258
Yung Pen	108	48	-2.7	47	21	3.2	70	31	2.0	225
The Sirens	118	56	-1.1	0	0	-5.3	92	44	5.7	210
Princess Bay	76	36	-4.9	14	7	-2.7	123	58	9.8	213
Palmer Head	117	50	-2.4	53	23	4.0	64	27	0.9	234
Flax Bay	78	100	4.2	0	0	-3.2	0	0	-4.4	78
Breaker Bay	169	60	-0.4	43	15	1.0	68	24	-0.1	280
Barrett Reef	190	76	2.7	29	12	-0.7	32	13	-3.7	251

Figure 2.4. Marginal means ($\pm$ SE) plots of temporal (seasonal, interannual) and spatial trends in water parameters for Wellington south coast's shallow nearshore zone (mean depth = 9.1m). Interactions and main effect shown were significant at $p = 0.05$. Mean seasonal temperatures did not differ by more than 1.6°C between the two survey years. The maximum observed seasonal difference did not exceed 4.5°C for each year. Salinity values were stable across seasons and between survey years (<1 psu difference). Mean seasonal values did show a small interaction with "Site", with winter values lower at the eastern end of the shore (Barrett Reef) but this difference was less than 2 psu. Temporal trends in dissolved oxygen values followed the same general pattern for year and season (highest values observed in the coldest season; lowest values in the warmer season). The mean difference between the two survey years was 0.2 mg/L and less than 1.5 mg/L among seasons for both years.

Figure 2.5 Marginal means plots ($\pm$ SE) of temporal (seasonal, interannual) and spatial trends in water clarity and seabed topography in Wellington south coast’s shallow nearshore zone (mean depth = 9.1m). Interactions and main effect shown were significant at $p = 0.05$. Mean Secchi depth differed seasonally and between survey years (2007/8 and 2008/09). Mean horizontal Secchi distances differed between seasons, with the lowest visibility observed in winter. A spatial trend in horizontal and vertical Secchi values identified the lowest water clarity at the eastern harbour entrance and highest clarity along the western and central sites. Horizontal Secchi values, measured closer inshore than vertical Secchi readings, identified lower clarity in these high subtidal sites. Rugosity, (seabed topography), differed only at the central “The Sirens” and the eastern “Flax Bay” sites.

Figure 2.6. The seabed composition of Wellington south coast’s nearshore zone (4-13.6m survey range). The physical habitat assessment was included in two annual pre-reserve baseline surveys of the epibenthos and conducted using the point-intercept method (n = 600). Substratum classes were defined using Wentworth’s (1922) classification scheme. *Reef orientation was logged for reefs higher than 1 metre and only during the second survey (n = 300). Surveys were conducted in 2007/08 and 2008/09.

Chapter 3. The ecological structure of Wellington south coast's shallow rocky-reef epibenthos

A Monster of the Abyss

1. Introduction

1.1 General introduction and background

Marine assemblages are characterised by natural temporal and spatial variation in community structure and processes (Andrew and Mapstone 1987, Dayton et al. 1998, Cole 2003). Surveys are an essential tool in quantifying the scales at which temporal and spatial patterns occur. Pre-impact baseline surveys provide background point-in-time descriptions of seascape structure and status, while replication in time and space increases the ability to identify the nature (positive, negative) and scale of human impacts on subtidal communities (Underwood 1994, Kingsford and Battershill 1998).

Ecosystem descriptors from baseline studies are used to develop remediation or conservation strategies whose success can then be assessed through subsequent targeted monitoring surveys. These surveys are most effective when their design and goals reference pre-existing baseline results, which can help to refine the process of selecting target species and field methods appropriate to an area's biotic and abiotic characteristics (McCrone 2001). Time, logistics, cost and safety issues are significant factors in the design and implementation of subtidal surveys, and the need to monitor a national network of marine reserves to a consistent standard calls for standardised field methods that can provide quickly obtained and accurate representative data (McCrone 2001, Anderson 2002, Cole 2003). Being mensurative, baseline surveys do not explain the mechanisms by which physico-chemical processes interact with each other and with the biotic component of an ecosystem (Dayton 1985a). However, the patterns and processes identified by baseline surveys can be used to develop hypotheses-testing manipulative experiments that explore biological and physical processes, natural and human impacts, and interactions among processes and impacts (Kingsford and Battershill 1998, Underwood and Chapman 2000). Understanding such interactions is essential for system-based resource management approaches (Baskett et al. 2007).

The choice of what to assess in a baseline survey is not always clear, although the interests (recreational, commercial, cultural, historical, custodial) in and concerns (impact effects) for a specific area can suggest a species (or group) or specific habitat

type to assess (Anderson 2002, Cole 2003). Common metrics of community structure include species identity, density, size-frequency classes, and biomass (Kingsford and Battershill 1998). Although an abundance survey of one or a few impacted species such as the sea urchin *Evechinus chloroticus* (kina) and the abalone *Haliotis iris* (paua) is quickly completed and provides useful information, it will not identify larger-scale changes in community composition, structural complexity, or interactive effects such as shifts in trophic structure due to the cascade effect of increased abundance of top predators (Halpern and Warner 2002). Baseline work can provide information on the biotic and abiotic components and processes of seascapes by quantifying: diversity, rarity, naturalness, ecosystem functioning, population size (abundance) and structure (size classes), productivity (trophic structure), uniqueness, special attributes, distribution and range, representativeness, vulnerability, substratum type, light, water movement, physico-chemical parameters, natural and human threat potential, and economic interest (Davies et al. 2001, Pomeroy et al. 2004). Formats in which survey output can be presented visually or manipulated include biological valuation mapping schemes (Deros et al. 2007), and planning and management software such as MARXAN (Ball et al. 2009).

Two approaches to selecting target species or groups for surveys use functional groups or focal species. Functional groups can be defined on the basis of morphology, physiology, motility, or trophic status, but classifications are not definitive and are driven by the research question (Littler and Littler 1984, Steneck and Dethier 1994, Mann 2000). Steneck and Dethier (1994) found that reducing structural redundancy by scaling up from macro-algal species identity to functional identity provided a more stable overview of community structure information than did taxonomy-defined analyses. However, Sanders et al. (2007) obtained an opposite result by multivariate analyses of benthic invertebrate biota: in that study, site groupings from multidimensional scaling techniques varied with the functional classification used. An advantage of using functional groups based on relatively few morphological traits over taxonomic identification requiring familiarity with specialized terms is that inexperienced divers will be able to easily collect accurate data (Bell 2007).

The definitions and applications of the collective term “focal species” and its component types have been widely discussed (Power et al. 1996, Zacharias and Roff

2001, Davic 2003). Component concepts such as “keystone”, “indicator”, “umbrella”, and “flagship” are seen as useful conservation and management tools for their ability to represent through one or a few species some aspect of community status (Davies et al. 2001, Pomeroy et al. 2004) (see chapter 1 for a more complete discussion). Advantages of baseline or monitoring surveys targeting a category of focal species include shorter time underwater and, again, the ability for volunteer and inexperienced surveyors to accurately assess a reduced number of variables. Although trained field volunteers can obtain survey estimates that are in high agreement with those made by professionals (Hassell et al. 2013), training volunteers can be a time-consuming process. In this study's experience, volunteer divers' abilities to produce low-error and –bias estimates can be limited by their specific interests and diving comfort in harsh field conditions. Any reduction of criteria to be assessed underwater by relatively untrained workers (often the case with dived surveys) increases reliability of the data. Despite these advantages, clearly the most comprehensive assessment possible will capture the most informative view of natural and anthropogenic patterns and changes for stakeholders (Anderson 2002).

1.2 Local practices and surveys

A 2001 review of New Zealand's use of baseline and monitoring surveys for assessing areas with reserve status or being considered for protection found that only half of the then-existing marine reserves (MRs) had been surveyed for baseline status before being gazetted. Of those baselines, the field methodologies and results of less than half had been incorporated into subsequent monitoring surveys (McCrone 2001). A lack of continuity between the two types of surveys and their datasets, when available, and lack of standardised field and archival protocols are impediments to the successful implementation and monitoring of the national marine reserve network being developed in accordance with New Zealand's Biodiversity Strategy (McCrone 2001, Anderson 2002, Department of Conservation and Ministry of Fisheries 2005). In the greater Wellington area, Oliver and Milne (2012) noted that coastal management of unprotected as well as protected areas would benefit from standardised field and data storage protocols and continuity among surveys.

1.3 Previous studies on WSC shallow rocky-reef community structure

Three surveys of Wellington south coast's (WSC) shallow rocky-reef communities have been conducted along sections of its ~ 14km range to provide resource managers with information on community structure and biodiversity. The first marine reserve-related study was conducted at the reserve's conception to assess the status of mobile macro-invertebrate and fish species of recreational interest along a portion of WSC (Owhiro to Houghton Bays) (Cole and Jackson, 1989). This followed an earlier observational study of benthic species at central and eastern sites that identified high levels of patchiness for two key mobile macro-invertebrate species (Choat and Schiel 1982).

A second reserve-related study, and the first large-scale baseline study for the purpose of pre-reserve benthic resource assessment was conducted by Pande (2000) at eight sites along a 12km stretch of coastline. Her study targeted five macro-algal canopy species characteristic of rocky-reef kelp forests, four recreationally important mobile macro-invertebrate species and 24 benthic and demersal fish species impacted by commercial and recreational fishing. The three-year seasonal survey at eight sites identified a spatial gradient in macro-algal and mobile macro-invertebrate abundances. Pande's (2000) recommendations for future baseline surveys included incorporating abiotic variables to identify the environmental drivers of the east-west gradient identified. One outcome of this study was the description of an east-west gradient in macro-algal and mobile macro-invertebrate abundances. Her study theorized that biotic and environmental factors might be responsible: a possible negative relationship between two canopy species, *Ecklonia radiata* and *Carpophyllum maschalocarpum*, and the classification of Barrett Reef as "exposed".

A later survey of WSC's shallow rocky-reef community for conservation purposes was by Shears and Babcock (2007). Their study included a broader range of epibenthic species than Pande's at 6 sites in the middle and eastern areas of WSC. One central (The Sirens) and one eastern site (Palmer Head) corresponded with two of Pande's locations. Their study used different field methods to identify depth-related patterns of all macro-algal and mobile and sessile macro-invertebrates present in their sampling area. Their community structure analyses also incorporated five

environmental variables. Results were used to develop a national biogeographic classification scheme of New Zealand's nearshore ecosystems (Shears et al. 2008).

1.4 Chapter aims

The Taputeranga Marine Reserve (green boundary, Figure 3.1) was gazetted in August 2008. Although the purpose of New Zealand marine reserves are to facilitate scientific research, interest in and support for a reserve along WSC was provided by a range of recreational and commercial stakeholders seeking to preserve a representative area of WSC subtidal seascape for the use and enjoyment of residents and visitors (Gardner, 2008). This conservation goal corresponds to the two roles of marine reserves: to protect unique or representative ecosystems and contribute to a network protecting the biological attributes of New Zealand's exclusive economic zone (Department of Conservation and Ministry of Fisheries 2005, Gardner et al. 2008).

My primary goal was to explore the biological structure of WSC's shallow rocky-reef epibenthic ecosystem. Specifically, I aimed to quantify the biotic component of WSC's shallow rocky-reef epibenthic ecosystem and the spatial scales at which variability occurs. I also aimed to identify the taxa responsible for spatial patterns and model the contribution of environmental factors to these patterns. By updating and expanding the scope of the previous baseline survey (Pande, 2000), these surveys will provide the first comprehensive baseline survey of biotic and abiotic components and their interactions along the full 14km extent of WSC. The data sets developed here will, with Pande's (2000) study, create the first long-term benchmark baseline data set in New Zealand for use in two aspects of resource management: 1) long-term evaluation of naturally occurring trends in a coastal ecosystem located off an expanding major urban centre; and 2) assessment of the impacts of reserve status. A secondary goal was to provide recommendations for post-reserve monitoring practices, including a set of representative species for expedited monitoring.

2. Methods

2.1 Field methods

2.1.1 Site selection and location

A comprehensive overview of Wellington south coast's (WSC) environmental and biological characteristics was presented in Chapter One. Two visual baseline surveys of rocky-reef communities were conducted during daylight hours and austral summers (late December-March) annually in 2008 and 2009. Sites from the previous south-coast survey (Pande, 1998-2000) were used here for continuity of data that would provide the archival basis of a long-term monitoring data set.

The final boundaries of the proposed Taputeranga Marine Reserve had shifted from those around which Pande's survey sites had been selected so a ninth site, referred to as 2nd Wash, was added at the western end to the eight original sites for a fully balanced model with three sites west of the proposed marine reserve (2nd Wash, 1st Wash, Red Rocks), three sites in the central or designated marine reserve area (Yung Pen, The Sirens, Princess Bay), and three sites east of the central/proposed marine reserve (Palmer Head/Gibraltar Rock, Flax/Breaker Bay, and Barrett Reef). This design set up a "beyond BACI (before/after/control/impact)" survey design with multiple control sites (three western, three eastern) for use in evaluating reserve effects in future monitoring studies (Andrew and Mapstone 1987, Underwood 1994). Site coordinates and location in relation to the shore are presented in Table 3.1. The central area shares the same final boundaries as those of Taputeranga Marine Reserve. Because the reserve gazetting date of August 2008 overlaps with this study's second annual baseline survey, the term "central" will be used here to avoid confusing these two baseline surveys with a reserve-effect study, which would necessarily take place after a passage of time to allow for a reserve impact on community structure to develop.

One of Pande's original eastern sites, Breaker Bay (Split Rock), was replaced with an alternative site that was more accessible, Flax Bay, during the first survey (2008) due to undiveable conditions at Breaker Bay during the survey season. Although Flax Bay physical habitat consists primarily of mobile substrata (boulder) with minimal bedrock, it was the only site near Split Rock that could serve as an alternate site. Both Flax and Breaker Bays were surveyed the second year to maintain continuity of data between past and future surveys at the original Breaker Bay site and

continuity with the previous year's Flax Bay data. The depth range of all surveyed sites, 5-12m was identified as having high biodiversity and abundance values for shallow northern rocky-reefs (Schiel 1988) and for not having depth-related assemblage shifts within this stratum (Choat and Schiel 1982).

Other modifications included adopting annual surveys over seasonal surveys due to the difficulty Pande encountered of completing four clearly-defined seasonal surveys each year within Wellington's limited number of diveable days. Additionally, the fish population component of her survey was deemed a separate survey for this study so was completed as a component of another project (Eddy 2010). Although video/still images create a visual database that can be used for quality control and archival referencing (Davies et al. 2001), this method of surveying, used by Pande to complement her visual censuses, and trialled during preliminary dives for this set of surveys, was discontinued. Limited seabed visibility, characteristic of the south coast, and kelp forest canopy cover constantly obscured understorey, turfing, and encrusting biota, making it difficult to obtain consistently high-quality video footage.

Table 3.1. Baseline survey site coordinates (waypoints) and values for environmental variables along Wellington, New Zealand's south coast. At each site, transects were laid from two starting hubs: "A" and "B". "Distance offshore" was measured perpendicular to the coast. "Distance from harbour" was measured from the harbour's front lead light. *Split Rock (Breaker Bay) was surveyed only during the second year. Annual surveys were conducted over two summers in 2008-09.

Site, Area (west to east)	Latitude south	Longitude east	Distance between hubs (m)	Distance to next waypoint (m)	Distance offshore (m) (Hub: A, B)	Distance from harbour (km)	Rugosity	Exposure	Mean depth (m)
2nd Wash A, West	-41.35764	174.70032					1.5	24	8.5
2nd Wash B, West	-41.35815	174.70229	180	830	128, 190	14.2			
1st Wash A, West	-41.36233	174.71272					1.3	21	9.5
1st Wash B, West	-41.36081	174.71183	180	1190	142, 178	13			
Red Rocks A, West	-41.35963	174.72446					1.6	20	7.9
Red Rocks B, West	-41.36035	174.72259	170	2895	139, 204	12			
Yung Pen A, Central	-41.34995	174.75391					1.5	13	6.9
Yung Pen B, Central	-41.35053	174.75443	100	795	83, 184	9			
The Sirens A, Central	-41.34970	174.76410					2.3	12	7.3
The Sirens B Central	-41.34948	174.76549	128	2389	94, 50	7.5			
Princess Bay A, Central	-41.34643	174.78822					1.5	2	6.1
Princess Bay B, Central	-41.34664	174.78876	53	3983	59, 79	6.8			
Palmer Head A, East	-41.34682	174.82183					1.4	2	9.2
Palmer Head B, East	-41.34654	171.82296	156	1104	174, 210	3.4			
Flax Bay A, East	-41.33605	174.82565					1.1	0	5.8
Flax Bay B, East	-41.33670	174.82547	80	896	83, 54	3			
Split Rock A*, East	-41.33289	174.83087					1.4	1	7.0
Split Rock B*, East	-41.33283	174.83197	95	739	121, 144	2.1			
Barrett Reef A, East	-41.34059	174.83568					1.6	2	8.1
Barrett Reef B, East	-41.34192	174.83513	153		911, 961	2.8

Figure 3.1. Baseline survey sites in relation to local human geography. Geography is characterised primarily by undeveloped land to the west (streams = blue lines) and the built environment extending from the centre eastward (sewage and stormwater infrastructure = purple lines, red circles). Baseline survey sites fall into three areas: "West", "Central", and "East". Sewer and stormwater data owned by Wellington City Council.

2.1.2 Species surveyed

Pande's previous full-scale survey focused on five macro-algal species and four mobile macro-invertebrate species of recreational and commercial interest. These were the native species *Carpophyllum maschalocarpum*, *Ecklonia radiata*, *Lessonia variegata*, *Macrocystis pyrifera* and the invasive *Undaria pinnatifida*. Mobile macro-invertebrate species included were *Evechinus chloroticus* (kina/sea urchin), *Haliotis australis* (yellowfoot paua/abalone), *H. iris* (blackfoot paua), and *Jasus edwardsii* (rock lobster).

To achieve the study's aims, the current survey's scope was expanded from the above species list to include all macro-algal, epibenthic mobile macro-invertebrate and epibenthic sessile macro-invertebrate species observed in quadrats and transects. The size of individuals which could be detected without displacing flora or substrata was ~3cm+. Species/functional groups and habitats that were not included due either to their absence during daytime sampling, the need for more specialised sampling techniques, or the time limitations imposed by scuba diving, included marine worms, hermit crabs, prawns, epizoa, non-emergent (juvenile) paua, and under-surfaces of mobile substrata. The final survey list comprised 47 macro-algal species or functional groups (Table 3.2), 36 mobile macro-invertebrate groups (Table 3.3), and 29 sessile macro-invertebrates identified to morphotype. These included 4 members of Ascidiacea, 7 Bryozoa, 2 Hydrozoa, and 15 Porifera (voucher images, Appendix A). Tentative identifications of sessile macro-invertebrates were based on Wing, 2008 and de Cook, 2010. Due to lack of available taxonomic expertise at the time of these surveys, individuals/colonies within each group were classified by morphotype as a valid proxy for taxonomic identification (Bell 2007).

Two macro-algae were identified to functional group: "geniculate coralline algae", which included *Arthrocardia*, *Cheilosporum*, *Corallina*, *Haliptilon*, and *Jania* species, and "crustose coralline algae", which included encrusting, warty, foliose and flat discoid species (Harvey et al. 2005). Eight Rhodophyta and five Phaeophyta were not identified (voucher images, Appendix B).

2.1.3 Survey methods

A visual census technique based on Pande's (2000) approach was slightly modified for these more comprehensive surveys. To assess the different structural groups and morphotypes present in the epibenthic community, abundances were measured using counts, percent cover, or both within a 1-m² folding quadrat with scale bars marked in 1cm increments (Murray 2001). Percent cover of macro-algal canopy was visually assessed by measuring the area covered by the canopy against the quadrat's scale bars when the plant was vertical in the water column. Area was then converted to the nearest 1% coverage. To assess percent cover of smaller colonial algae and sessile macro-invertebrates, a 0.01m² quadrat was used. Mobile macro-invertebrates were surveyed over four 2 x 25 metres belt transects (200m² per site). Macro-algae and sessile macro-invertebrates were assessed at ten 1-m² quadrats per site (10m² per site). To better capture patterns of abundance for patchily distributed species, the transect width was doubled for total transect dimensions of 4 x 25 metres (400 m² per site) during the second annual survey. This width was selected because volunteer divers could easily gauge the boundaries visually using the central transect tape as a reference line. The original transect length of 25 metres was retained because longer pilot transects often extended beyond reef platforms into soft-bottom communities.

A visual census technique was used for assessing sessile macro-invertebrates because this method has been reported as more accurate than random-point quadrats (Dethier et al. 1993). To ensure independence of sampling units (quadrats or transects), at each site two starting hubs for transects were haphazardly selected on the basis of that day's weather, tide, current and anchoring conditions. Hub waypoints were logged and were located far enough apart to ensure that transects did not cross, but were close enough to be considered at the same site (mean hub distance = 66.7m) (see Table 3.1 for site distribution details). At each hub, two belt transects were deployed, with one set of 5 randomly placed quadrats distributed along one of the transects. Transect headings were chosen from a randomly-generated list of numbers, with unique lists used for each hub, site and year. Headings were rounded to 10° intervals for ease of diver navigation in the strong surges characteristic of south coast dive sites. Transect widths were measured using metre-sticks. If a transect ran off the reef platform at less than half the transect length (12.5m), an arbitrarily selected

distance, then a second transect was re-laid using a different random heading. Different randomly-generated numbers were used for each hub, site and year.

Table 3.1. Taxonomic list of macro-algae observed during two annual baseline surveys of Wellington, New Zealand's south coast rocky-reef epibenthic communities (5-12m) (austral summers, 2007/8 & 2008/9). In addition, 3 unidentified understorey Phaeophyta and 8 unidentified understorey Rhodophyta were logged during the surveys. Percent cover was assessed for all species. * indicates species that were also counted.

SPECIES	FAMILY	ORDER	CLASS	Phylum
<i>Asparagopsis armata</i> Harvey	Bonnemaisoniaceae	Bonnemaisoniales	Florideophyceae	Rhodophyta
<i>Carpomitra costata</i> (Stackhouse) Batters	Sporochneaceae	Sporochneales	Phaeophyceae	Phaeophyta
<i>Carpophyllum flexuosum</i> * (Esper) Greville	Sargassaceae	Fucales	Phaeophyceae	Phaeophyta
<i>Carpophyllum maschalocarpum</i> * (Turner) Greville	Sargassaceae	Fucales	Phaeophyceae	Phaeophyta
<i>Caulerpa brownii</i> (C. Agardh) Endlicher	Caulerpacaeae	Caulerpales	Bryopsidophyceae	Chlorophyta
<i>Caulerpa flexilis</i> Lamouroux	Caulerpacaeae	Caulerpales	Bryopsidophyceae	Chlorophyta
<i>Caulerpa geminata</i> Harvey	Caulerpacaeae	Caulerpales	Bryopsidophyceae	Chlorophyta
<i>Enteromorpha intestinalis</i> (Linnaeus) Nees	Cladophoraceae	Cladophorales	Ulvophyceae	Chlorophyta
<i>Champia laingii</i> Lindauer	Champiaceae	Rhodymeniales	Florideophyceae	Rhodophyta
<i>Cladhymania oblongifolia</i> Harvey	Rhodomelaceae	Ceramiales	Florideophyceae	Rhodophyta
<i>Colpomenia sinuosa</i> (Roth) Derbes et Solier	Scytosiphonaceae	Scytosiphonales	Phaeophyceae	Phaeophyta
<i>Cystophora scalaris</i> * J. Agardh	Cystoseiraceae	Fucales	Phaeophyceae	Phaeophyta
<i>Desmarestia ligulata</i> (Lightfoot) Lamouroux	Desmarestiaceae	Desmarestiales	Phaeophyceae	Phaeophyta
<i>Ecklonia radiata</i> * (C. Agardh) J. Agardh	Alariaceae	Laminariales	Phaeophyceae	Phaeophyta
<i>Eupilota formosissima</i> (Montagne) Kuetzing	Ceramiales	Ceramiales	Florideophyceae	Rhodophyta
<i>Gigartina circuncincta</i> J. Agardh	Gigartinaceae	Gigartinales	Florideophyceae	Rhodophyta
<i>Gigartina decipiens</i> (Hook. F. et Harvey) Hook. f. et Harvey	Gigartinaceae	Gigartinales	Florideophyceae	Rhodophyta
<i>Glossophora kunthii</i> * (C. Agardh) J. Agardh	Dictyotaceae	Dictyotales	Phaeophyceae	Phaeophyta
<i>Halopteris</i> sp. not identified reliably to species	Stypocaulaceae	Sphacelariales	Phaeophyceae	Phaeophyta
<i>Landsburgia quercifolia</i> * (Hook. f. et Harvey) Harvey)	Cystoseiraceae	Fucales	Phaeophyceae	Phaeophyta
<i>Lessonia variegata</i> * J. Agardh	Lessoniaceae	Laminariales	Phaeophyceae	Phaeophyta
<i>Macrocystis pyrifera</i> * (Linnaeus) C. Agardh	Alariaceae	Laminariales	Phaeophyceae	Phaeophyta
<i>Margarinella</i> spp. (boryana, urvillia) * (A. Richard) Tandy	Seirococcaceae	Fucales	Phaeophyceae	Phaeophyta
<i>Melanthalia absissa</i> (Turner) Hook. f. et Harvey	Gracilariaceae	Gracilariales	Florideophyceae	Rhodophyta
<i>Plocamium microcladioides</i> South et Adams	Plocamiaceae	Gigartinales	Florideophyceae	Rhodophyta
<i>Pterocladia lucida</i> (Turner) J. Agardh	Gelidiaceae	Gelidiales	Florideophyceae	Rhodophyta
<i>Rhodymenia dichotoma</i> Hook. f. et Harvey	Rhodymeniaceae	Rhodymeniales	Florideophyceae	Rhodophyta
<i>Sargassum sinclairii</i> * Hook. F. et Harvey	Sargassaceae	Fucales	Phaeophyceae	Phaeophyta
<i>Ulva lactuca</i> Linnaeus	Ulvaceae	Ulvales	Ulvophyceae	Chlorophyta
<i>Undaria pinnatifida</i> (Harvey) Suringar	Alariaceae	Laminariales	Phaeophyceae	Phaeophyta
<i>Xiphophora gladiata</i> (Labillardière) Montagne	Fuaceae	Fucales	Phaeophyceae	Phaeophyta
<i>Zonaria turneriana</i> J. Agardh	Dictyotaceae	Dictyotales	Phaeophyceae	Phaeophyta
Encrusting coralline algae (group): Morphotypes included: encrusting, warty, foliose, flat discoid.				Rhodophyta
Geniculate coralline algae (group): genera identified included: <i>Arthrocardia</i> , <i>Cheilosporum</i> , <i>Corallina</i> , <i>Haliptilon</i> , <i>Jania</i>				Rhodophyta

Table 3.2. Taxonomic list of macro-invertebrate species observed and assessed during two annual baseline surveys of Wellington, New Zealand's south coast rocky-reef epibenthic communities (5-12m) (austral summers, 2007/8 & 2008/9). Count data were logged for all species.

Species	Family	Order	Class	Phylum
<i>Aphelodoris luctuosa</i> Cheeseman, 1882	Discodorididae	Doridoidea	Gastropoda	Mollusca
<i>Archidoris wellingtoniensis</i> Abraham, 1877	Archidorididae	Nudibranchia	Gastropoda	Mollusca
<i>Argobuccinum pustulosum tumidum</i> Lightfoot, 1786	Ranellidae	Neotaenioglossa	Gastropoda	Mollusca
<i>Astrostele scabra</i> Farguharson, 1894	Asteriidae	Forcipulatida	Asteroidea	Echinodermata
<i>Buccinulum linea</i> Martyn, 1784	Buccinidae	Neogastropoda	Gastropoda	Mollusca
<i>Cabestana spengleri</i> Perry, 1811	Ranellidae	Neotaenioglossa	Gastropoda	Mollusca
<i>Cantharidus opalus</i> Martyn, 1784	Trochidae	Vetigastropoda	Gastropoda	Mollusca
<i>Cantharidus purpureus</i> Gmelin, 1791	Trochidae	Vetigastropoda	Gastropoda	Mollusca
<i>Chromodoris aureomarginata</i> Cheeseman, 1881	Chromodorididae	Nudibranchia	Gastropoda	Mollusca
<i>Cominella virgata</i> H. Adams & A. Adams, 1853	Buccinidae	Neogastropoda	Gastropoda	Mollusca
<i>Cookia sulcata</i> Gmelin, 1791	Turbinidae	Vetigastropoda	Gastropoda	Mollusca
<i>Coscinasterias calamaria</i> Gray, 1840	Asteriidae	Forcipulatida	Asteroidea	Echinodermata
<i>Diplodontias dilatatus</i> Perrier, 1875	Odontasteridae	Valvatida	Asteroidea	Echinodermata
<i>Evechinus chloroticus</i> Valenciennes, 1846	Echinometridae	Echinoidea	Echinoidea	Echinodermata
<i>Haliotis australis</i> Gmelin, 1791	Haliotidae	Haliotoidea	Gastropoda	Mollusca
<i>Haliotis iris</i> Gmelin, 1791	Haliotidae	Haliotoidea	Gastropoda	Mollusca
<i>Haustrum haustorium</i> Gmelin, 1791	Muricide	Neogastropoda	Gastropoda	Mollusca
<i>Ischnochiton maorianus</i> Iredale, 1914	Ischnochitonidae	Ischnochitonina	Ampinea	Mollusca
<i>Josus edwardsii</i> Hutton, 1875	Palinuridae	Decapoda	Malacostraca	Arthropoda
<i>Maoricolpus roseus</i> Quoy and Gaimard, 1834	Turritellidae	Caenogastropoda	Gastropoda	Mollusca
<i>Modolia granulosa</i> Martyn, 1784	Turbinidae	Vetigastropoda	Gastropoda	Mollusca
<i>Octopus vulgaris</i> Lamarck, 1798	Octopodidae	Octopoda	Cephalopoda	Mollusca
<i>Patiriella regularis</i> Verrill, 1867	Asterinidae	Valvatida	Asteroidea	Echinodermata
<i>Pectinura maculata</i> Verrill, 1869	Ophiodermatidae	Ophiurida	Ophiuroidea	Echinodermata
<i>Penion sulcatus</i> Lamarck, 1816	Buccinidae	Neogastropoda	Gastropoda	Mollusca
<i>Pentagonaster pulchellus</i> Gray, 1840	Pentagonasterinae	Valvatida	Asteroidea	Echinodermata
<i>Phlyctenactis tuberculosa</i> Quoy and Gaimard, 1833	Actiniidae	Actinaria	Anthozoa	Cnidaria
<i>Plagusia chabrus</i> Linnaeus, 1758	Grapsidae	Decapoda	Malacostraca	Arthropoda
<i>Plaxiphora obtecta</i> Pilsbry, 1893	Mopaliidae	Ischnochitonina	Ampinea	Mollusca
<i>Scutus breviculus</i> Blainville, 1817	Fissurellidae	Archaeogastropoda	Gastropoda	Mollusca
<i>Stichaster australis</i> Verrill, 1817	Stichasteridae	Forcipulatida	Asteroidea	Echinodermata
<i>Stichopus mollis</i> Hutton, 1872	Stichopodidae	Aspidochiroidea	Holothuroidea	Echinodermata
<i>Stegaster inflatus</i> Hutton, 1872	Echinasteridae	Spinulosa	Asteroidea	Echinodermata
<i>Sypharochiton pelliserpentis</i> Quoy & Gaimard, 1835	Chitonidae	Ischnochitonina	Ampinea	Mollusca
<i>Trochus viridis</i> Gmelin, 1791	Trochidae	Vetigastropoda	Gastropoda	Mollusca
<i>Turbo smaragdus</i> Gmelin, 1791	Turbinidae	Vetigastropoda	Gastropoda	Mollusca

2.1.4 Community metrics selected for survey

Abundance was quantified with count and percent cover information as discussed above. Identification was made to species level to provide the greatest level of taxonomic detail for current and future reference. This level was also chosen because it allows scaling up to functional group of choice as desired for future analyses. Finally, it provides continuity among past and future data sets. Mobile macro-invertebrates were counted and three species of commercial and recreational importance were measured using rulers to the nearest 5mm: *Evechinus chloroticus* (kina/sea urchin: test diameter), *Haliotis australis* (yellowfoot paua/abalone: shell length), and *H. iris* (blackfoot paua: shell length). Additionally, *Jasus edwardsii* (rock lobster) was assessed as “below legal size” (female < 54mm; male < 60mm tail width) or “legal”. This option was chosen to avoid the damage caused by manually capturing individuals for measuring. To provide a common metric for the entire macro-algal community, which included colonial-type morphologically indistinct structures, percent cover m^{-2} of all species was measured. Because percent cover can change rapidly due to physical (storm surges, sweeping) and biological (grazing, disease) processes, more robust estimates of abundance, counts, were collected for eleven structurally discrete canopy species (marked with asterisk in Table 3.2) (Levine 1985).

Total plant lengths of seven canopy species were also measured to the nearest centimetre: *Carpophyllum flexuosum*, *C. maschalocarpum*, *Ecklonia radiata*, *Landsburgia quercifolia*, *Lessonia variegata*, *Macrocystis pyrifera*, and *Marginariella boryana* (the latter only in 2009 due to species misidentification with *M. urvilliana* in 2008). Details of measurement methods are described in chapter 5. Sessile macro-invertebrates in each quadrat were visually estimated by percent cover m^{-2} to the nearest 0.01%.

Physical habitat associations for macro-algae, mobile and sessile macro-invertebrates were recorded at each quadrat and at meter intervals for belt transects. Associations between macro-invertebrates and host macro-algae were also logged for a related study. For a full description of identification and classification methods, see chapter 2. For a brief review, see the next section.

2.1.5 Environmental features and processes

Environmental features recorded on each survey dive were: depth (from dive computers), habitat, and sediment. Depths were not adjusted for tidal phase because tide tables were only available for the eastern end of the south coast (harbour entrance) and times were unpredictable at the western sites. However, mean maximum tidal range (1.7m) was relatively low. Habitat class analysis and categorization were discussed in detail in Chapter 2. To briefly review, classes were defined by percent of substratum observed to yield five classes, each dominated by the first component listed. "Mixed" indicates smaller size classes present in varying proportions totalling less than 50%. This yielded five classes: "Sand/Pebble"; "Cobble, mixed"; "Boulder, mixed"; "Broken reef, mixed", and "Reef". "Broken reef, mixed" included reef edges and transitions between patchy bedrock (size: boulder < patchy bedrock < 5m diameter reef) and mobile substrata ranging from sand to mixed boulders. "Reef" was subdivided according to orientation: horizontal, inclined, vertical. "Inclined" refers to non-horizontal, sloping emergent bedrock while "vertical" refers to reef walls higher than 1 metre. This was an arbitrarily selected height based on observed patterns among south coast shallow-reefs. Reef orientation was recorded because it has been identified as structuring epibiotic macro-algal and sessile invertebrate rocky-reef assemblages and (Glasby and Connell 2001, Vaselli et al. 2008). Wave-pattern data was not available at the small spatial scale at which this study took place.

Sediment was assessed only for quadrats because the disturbance to the seabed caused by diver movements during belt transect deployment, sampling and measuring displaced sediment. Quadrats were therefore surveyed for sediment before abundance at each hub to capture an undisturbed estimate of this feature. Sediment was classified into "None", "Low (fine film present; organism/habitat features not obscured)", "Medium (film present, partial obscuring of organism/habitat features)", or "High (heavy film present, organism/habitat features fully covered/obscured)". Sediment was selected for inclusion because of its structuring effect on macro-algal and macro-invertebrate communities (Airolidi 1998, Lohrer et al. 2006, Phillips and Shima 2006).

Additional environmental factors included in analyses were: depth, temperature, dissolved oxygen, salinity, Secchi depth, rugosity, exposure, distance from harbour, and hydrology. Field sampling methods and mean values for rugosity and water quality (temperature, dissolved oxygen, salinity, Secchi depth) were presented in Chapter 2. Summer values for the four water quality variables were averaged by site and by year for analyses. Because 2nd Wash was not included in the water-quality sampling program, values for “1st Wash” were used for this site. These values were deemed a suitable substitute because of their proximity and their location west of the Karori Rip, a potential hydrological barrier to larval transport.

Mean depth refers to values averaged by year for each site. Exposure was calculated using a modified Baardseth method on a chart with a scale of 1: 100,000. This method sums the number of unobstructed 10⁰ sectors deeper than 5m over a 7.5km radius to produce site values ranging from “0”, or fully sheltered, to “36”, or fully exposed (Wernberg and Thomsen 2005) (Table 3.1).

“Distance from harbour” was measured from the harbour’s front lead light as an arbitrary starting point and followed the coastline at a distance offshore that corresponded to the survey sites’ offshore distances. This variable was included to model two possible processes affecting south coast water quality: pollutant run-off and suspended sediment transport. South coast contamination events typically occur at one or more sites annually and were discussed in chapter one’s site description. The Orongorongo and Wainuiomata Rivers east of the harbour entrance were identified as the major source of terrigenous sediment carried westward along the south coast (Grapes and Goh 2008).

To model the possible barrier effect of the Karori Rip on spatial differences in epibenthic community structure, “hydrology” was scaled as “yes” for the two sites west of the Karori rip (2nd Wash and 1st Wash) and “no” for the remaining 7 sites. The rip was identified as a possible structuring process because of its characteristic 5 knot/hour (2.5m/sec) surface currents (LINZ Chart 463) and because hydrology can structure benthic communities through sediment transport, deposition and water column mixing (Lamare 1998, Snelder et al. 2005). “Urban development” was selected as a variable to model the possible influence of urban infrastructure on the central and eastern sites (The Sirens, Princess Bay, Palmer Head, Flax Bay, Barrett Reef) versus the

rural scrubland of the western and central coastal areas (2nd Wash, 1st Wash, Red Rocks). This factor was also intended as a surrogate for on-going paua poaching anecdotally reported by local residents near the western sites

2.2 Statistical analyses

Due to the differences in sampling areas covered by the quadrat and belt transect methods, and to the use of difference measures of abundance to accommodate discrete and colonial taxa (counts versus percent cover), analyses were run separately on the three epibenthic groups. The macro-algal group was divided into 2 subsets, an 11-species group assessed by counts and the full macro-algal group assessed by percent cover. The other two groups were mobile macro-invertebrates, assessed by counts, and sessile macro-invertebrates (percent cover). The full macro-algal group and sessile macro-invertebrates were analysed separately because differences in ranges were large enough to obscure each group's patterns. Analysis of the complete epibenthic assemblage based on presence/absence data per site was not possible because this conversion left no replicates for analyses.

Analyses were run on species-level identification. Exceptions were: 1) *Marginariella boryana* and *M. urvillia*, which were entered as *Marginariella* spp. due to confounding the two species during the first survey; 2) *Caulerpa brownii* and *C. flexilis*, which were entered as *Caulerpa* spp. because the two often grew intermingled in patches; 3) *Halopteris* spp.; 4) the functional group "crustose coralline algae" (CCA) because identification of these species was beyond the scope of this project; and 5) "geniculate coralline algae" as described above. Unidentified taxa were included as "Unknown [Taxon] A", etc. The statistical packages SPSS, PRIMER 6, and PERMANOVA+ for PRIMER were used for analyses. All p-values were considered significant at $p < 0.05$.

To visualise multivariate patterns in assemblage structure in a reduced space, non-metric multi-dimensional scaling (MDS) plots from Bray-Curtis resemblance matrices were constructed. Count data were square-root transformed and percent cover data were treated as standardised (Anderson et al. 2008a). To simplify visualisation, ordinations were run on replicates averaged by "Site" within each level

of “Year”. These plots were visually compared with those run on full data sets to ensure that patterns in plots of averaged data adequately represented the patterns in full data sets. These plots also provided a visual interpretation of significant differences among sites identified by PERMANOVA (see below).

Spatial and temporal differences in assemblage structure were explored using PRIMER's PERMANOVA (permutational multivariate analysis of variance) routine. A two-way fixed factor design with interaction term was used, with factors “year” ($n = 2$) and “site” ($n = 10$). Permutational F-values were generated from 9999 randomisations. To complement PERMANOVA's tests for differences in location, PERMDISP (permutational test for dispersion) was used to test for differences in dispersion among levels of each factor. Note that significant differences in dispersion, however, do not preclude the use of PERMANOVA (Anderson et al. 2008b). The MDS plots, described above, provided visual interpretations of PERMANOVA results.

To identify which species explained similarity/dissimilarity in assemblage structure among sites, PRIMER's SIMPER routine was used. Because crustose coralline algae (CCA) dominated percent cover (mean = $73\% \text{ m}^{-2} \pm 2.4 \text{ SE}$), this group was excluded from analyses to allow detection of other variables' contribution to multivariate patterns. The contribution cut-off point was arbitrarily set at 80% (Anderson et al. 2008b). Analyses were run at the spatial scale of “area” to produce a manageable output. Levels of this factor are “west”, “central”, and “east”.

Patterns in size classes for three mobile macro-invertebrates and seven macro-algal canopy species were assessed visually using size-frequency plots. These species were selected due either to their role in local fishing (macro-invertebrates) or their status as either engineer canopy species or the most commonly occurring canopy species (macro-algae).

To identify differences in multivariate macro-algal assemblage patterns described by the measures “count” versus “percent cover”, PRIMER's non-parametric RELATE routine was used on an 11-species subset. This non-parametric Mantel test compares a similarity matrix against a ranked matrix to determine ranked correlations between count and percent cover data sets. A Bray-Curtis matrix of percent cover data against square-root transformed count data was used. Statistical significance was calculated from 9999 permutations (Clarke and Warwick 2001).

To identify subsets of species that could be used for expedited surveys in the future, PRIMER's BEST routine was run on each epibenthic group. The BEST routine finds high-rank correlations between resemblance matrices of subsets and full matrices to identify structural redundancies in species/group data sets. Results can be used to develop representative subsets of species that model full-set variability. BVStep options selected were "stepwise using Spearman rank correlation" with the rho criterion for "best subset" set > 0.95. Tests were run from both fixed starts (no variables) and random starts (randomly-selected seed variables) to check the robustness of the resulting subsets. Random restarts were set = 5 to avoid false solutions (Clarke and Warwick 1998).

Multivariate patterns in the environmental variable data set were visually presented in MDS plots from Bray-Curtis matrices of Euclidean distances on normalised data (Anderson et al. 2008a). To simplify patterns, ordinations were run on replicates averaged by "Site" within each level of "Year". These plots were visually compared with those run on full data sets to ensure that averaged plots adequately represented patterns defined in full data sets. Vector overlays of Spearman-ranked correlation values identified environmental contributions to MDS patterns.

To identify the contribution of the categorical and continuous environmental variables, discussed in section 2.1.5 above, to multivariate assemblage structures of each epibenthic group, the non-parametric DistLM (distance-based linear model) routine was used. This procedure partitions variation similarly to multiple linear regressions. The complementary dbRDA (distance-based redundancy analysis) was used to visually represent any significant relationships between environmental and biological matrices. These plots fit significant values from the DistLM regression in a constrained principle components ordination.

Analyses were run on Bray-Curtis measures of percent data. Percent data was zero-adjusted for the sessile macro-invertebrate group. Mobile macro-invertebrate data was dispersion weighted and square-root transformed for ranking in a Bray-Curtis resemblance matrix. Dispersion weighting is preferred over simple transformation because it selectively adjusts the contribution of consistently present species with patchy distribution so that other consistently present but less abundant species retain their influence in the multivariate structure. Using this technique with transformation

allows less abundant yet common species to contribute to the multivariate biological pattern (Clarke et al. 2006).

DistLM automatically standardises environmental variables during the routine. Euclidean distances were used for abiotic variables. The variable “sediment” was not included in the DistLM test of the mobile macro-invertebrate group because it was not possible to reliably estimate this over the large areas covered by belt transects. Variables were entered using a step-wise entry procedure and the most parsimonious model was selected using the Akaike Information Criterion (AIC). All tests were run for 9999 permutations and all p-values were obtained from these randomisations.

Difference in total number of species and in mean abundance m^{-2} among seabed classes were tested with one-way between-group analyses of variance (ANOVA) and Kruskal-Wallis tests. ANOVAs were used for “macro-algae by counts” and “macro-algae by percent cover” because these data met test assumptions of normality, homoscedasticity, and independence of residuals. Kruskal-Wallis tests were used for “sessile macro-invertebrates” and “mobile macro-invertebrates” because these data did not meet parametric test assumptions (Quinn and Keough 2006). Because the macro-algal abundance by counts data set included only a subset of all observed species, a test on total species numbers could not be run.

3. Results

Patterns in abundances (counts, percent cover) summarised by species and survey site for each survey year are presented by functional group in Figures 3.2-3.4. The species presented were identified by BESTStep as “indicator” species; taxa representing the biological characteristics of the epibenthic assemblages at each site (see section 3.2.2. below for discussion of this analysis). Abundance patterns for all groups were characterised by large spatial and interannual variation. Within each functional group, no single spatial pattern dominated. Following is a brief overview of the patterns, with subsequent sections presenting the results of analyses of assemblage structure using abundances and presence/absence data.

Mean abundances for macro-algal species identified spatial trends and showed less obvious interannual variation than observed in either macro-invertebrate group

(Figure 3.2). Abundances derived from count and percent cover estimates for six species did not differ markedly (Figure 3.2, top vs. bottom rows for *Carpophyllum flexuosum*, *C. maschalocarpum*, *Ecklonia radiata*, *Lessonia variegata*, *Marginariella* spp, *Macrocystis pyrifera*). Species with low abundances at central sites included *C. flexuosum*, *E. radiata*, *Marginariella* spp, *M. pyrifera*, and *Caulerpa* spp. Species with higher values at the western and/or eastern ends of WSC included *L. variegata* and *Marginariella* spp (west), and *M. pyrifera* (east). Geniculate coralline algae and *Halopteris* spp, and *Zonaria tureriana* abundances were higher at central sites. The invasive species *Undaria pinnatifida* was observed at only two sites, central Yung Pen and eastern Flax Bay. These were the two shallowest survey site.

Many of the mobile macro-invertebrate abundances followed non-linear spatial trends that differed among species, creating groups of species whose low/high abundances or absences corresponding to an area along WSC (western, central, eastern) (Figure 3.3). Mean abundances were higher at central sites for *Halitotis australis*, *H. iris*, *Plagusia chabris* and *Turbo smaragdus*. Species with higher values at western sites were *Evechinus chloroticus* and *Jasus edwardsii* and, for the eastern sites, *Patiriella regularis*, *Pectinura maculata*, *Pentagonaster pulchellus*, *Scutus breviculus*, and *Stichopus mollis*. Interannual variation at each site was low for most species, but large differences in means relative to interannual differences at other sites for the species, or absences during a survey year, were noted for the topshell *Cantharidus purpureus*, the abalone *H.iris*, the brittle star *P. maculata*, and the ducksbill limpet *S. breviculus*.

Sessile macro-invertebrate abundances were characterised by low values for all groups, large interannual variation among sites, and patchy distribution (Figure 3.4). Observed mean values were <1% percent cover m⁻² for all but Hydrozoa A. Only two ascidians were observed at all sites and no species were recorded at all sites in both years. Mean abundances for only four species, Porifera B and E and Ascidiacea A and D, were relatively evenly distributed among most sites, if not between years. The remaining sessile taxa were characterised by strong spatial patterns, with Bryozoa A, Hydrozoa B, and Porifera O more abundant at the western sites, Porifera M at western and central sites.

Figure 3.2. Mean abundances ($\pm$ SE) of macro-algal species identified by BEST BVStep analyses as representative of a larger set of Wellington's south coast nearshore rocky reef assemblages. Data were logged during two annual pre-reserve baseline surveys (summers, 2007/08 and 2008/09 (n = 10 quadrats). The site "Breaker Bay" was surveyed only in the second year. Count and percent cover data are presented for those species whose structure allowed discrete counts.

Figure 3.2. Mean abundances ($\pm$ SE) of macro-algal species identified by BEST BV5step analyses as representative of WSC benthos , page 2.

Figure 3.2. Mean abundances (± SE) of macro-algal species identified by BEST BVStep analyses as representative of WSC epibenthos, page 3.

Figure 3.3. Mean abundances (± SE) of mobile macro-invertebrate species identified by BEST BVStep analyses as representative of a larger set of Wellington's south coast nearshore rocky reef assemblages. Data were logged during two annual pre-reserve baseline surveys (summers, 2007/08 and 2008/09 (n = 4 transects). The site "Breaker Bay" was surveyed only in the second year.

Figure 3.3. Mean abundances (± SE) of mobile macro-invertebrate species, page 2.

Figure 3.3. Mean abundances ($\pm$ SE) of mobile macro-invertebrate species, page 3.

SITES: WEST TO EAST

Figure 3.4. Mean abundances (± SE) of sessile macro-invertebrate species identified by BEST BYStep analyses as representative of a larger set of Wellington's south coast nearshore rocky reef assemblages. Data were logged during two annual pre-reserve baseline surveys (summers, 2007/08 and 2008/09 (n = 10 quadrats). The site "Breaker Bay" was surveyed only in the second year. See Appendix A for voucher images.

Figure 3.4. Mean abundances ($\pm$ SE) of sessile macro-invertebrate species identified by BEST BVStep analyses as representative of WSC epibenthos, page 2. See Appendix A for voucher images.

SITES: WEST TO EAST

Figure 3.4. Mean abundances ($\pm$ SE) of sessile macro-invertebrate species identified by BEST BVStep analyses as representative of WSCepibenthos, page 3. See Appendix A for voucher images.

3.1 Aim 1: Describe and quantify the current status of the WSC shallow rocky-reef epibenthic communities

3.1.1 Overview of spatial and temporal patterns in assemblage structures by epibenthic group: MDS plots

MDS plots from abundance data for each epibenthic group identified distinct spatial and temporal (interannual) trends in assemblage composition, with distance among sites corresponding to differences in assemblage structure. Stress levels for all plots were equal to or below 0.17 (Figures 3.5, 3.6), indicating that plots adequately represented higher-order variation in assemblage structure (Clarke and Warwick 2001).

Within the macro-algal count data group, two clear patterns in assemblage structure emerged. First, the largest dissimilarity among sites was between Flax Bay and the remaining sites, with the similarity of these two groups of sites defined at only the 50% similarity level (Figure 3.5). Second, within the group of remaining sites, eastern sites clearly separated from central sites, with western sites falling between these two groups. Both these trends were also reflected in the macro-algal percent cover data, with the distance between eastern sites and the west/central sites even more pronounced. Here again, western and central sites were intermixed. However, only one of the two Flax Bay sites (first survey) separated from the remaining sites. Interannual differences at each site were clear for both count and percent cover data.

The mobile macro-invertebrate groups also showed two distinct spatial trends in assemblage composition, but among different sites than noted above (Figure 3.5). First, one group consisting of the westernmost sites (2nd Wash, 1st Wash) and the easternmost site (Barrett Reef) separated from the remaining sites. The third western site, Red Rocks, bridged the two groups, mirroring its actual location along WSC. Second, the remaining group comprised the central and eastern sites, but with a much clearer separation than noted in the macro-algal plots. These two groups were identified as only 50% similar. Interannual differences at each site were less pronounced for this data set than for macro-algae.

Sessile macro-invertebrate assemblages showed little similarity in composition at the 50% similarity level. As with the other two groups, the assemblage at Flax Bay during the first survey differed from all other sites for both surveys. The greatest dissimilarity was between the central sites and the western and eastern sites (Figure 3.5). Western sites here bridged the differences between the central sites and the easternmost site, Barrett Reef, similarly to the mobile macro-invertebrate pattern noted above.

Figure 3.5. Spatial and temporal patterns in assemblage structures of three epibenthic groups surveyed annually during summer WSC rocky-reef baseline surveys, 2008-09. Non-metric multi-dimensional ordinations of abundances were run on Bray-Curtis similarities of percentage cover and dispersion-weighted, square-root transformed counts. Stress is presented for each plot. Biological assemblages that are similar at the 50 % similarity level are indicated by the green boundaries. Site symbols are labelled by area along the south coast. Breaker Bay (East) was surveyed only during the second year.

Figure 3.6. Spatial and interannual variability in assemblage structure using presence/absence data. **A:** Local-scale patterns by site; **B:** larger-scale patterns by area. A linear spatial trend was suggested by the distance between most western and eastern sites.. Interannual differences were large for several sites in all three areas. Non-metric multi-dimensional ordinations of presence/absence data were run on Jacquard resemblance matrices. **Green boundaries** indicate biological assemblages that are similar at the 50 % similarity level. Surveys were conducted annually during summers, 2007/08 and 2008/09 on Wellington, New Zealand's south coast nearshore rocky-reef communities.

When defined on the basis of taxonomic identity alone (presence/absence) for all groups combined, patterns in assemblage composition differed from those for individual groups (Figure 3.6). Two distinct patterns emerged: first, two groups were again defined at the 50% similarity level. Unlike the site groupings for the individual groups defined by abundance, above, the first group here comprised three eastern sites, Breaker Bay, Barrett Reef, and Flax Bay (second survey). Similarity to the patterns above, however, within the second group western and central sites differed on a smaller scale, with the western Red Rocks site again bridging the distance between the two areas. Interannual differences at each site were larger than noted for abundance data.

3.1.2 Spatial and temporal differences in sites: PERMANOVA and PERMDISP

The PERMANOVA routine was used to test for significant differences in multivariate assemblages among sites and between years, while PERMDISP was used to test for significant differences in dispersion (variation) of data. Together, these two routines identified whether differences in levels of each factor were due to differences in location ("mean") or dispersion (multidimensional "variance").

Significant differences in location and dispersion were identified. Locations (distances to group centroids) differed for main factors and interactions in all groups (all $p < 0.005$, Table 3.4). Dispersions differed significantly for "Site" x "Year" interactions for macro-algae (counts and percent cover) and for the sessile macro-invertebrate group. Dispersion for the mobile macro-invertebrate group only differed significantly for "Year" (Table 3.5). Together, these differences in location and dispersion indicate that the differences in assemblage structure among sites vary within each year.

Post-hoc pairwise comparisons, a total of 82 per epibenthic group, for the interaction term identified 69 significant differences in sites for macro-algae, 64 for mobile macro-invertebrates, and 53 for sessile invertebrates (Appendix G). The shorter list of non-significant results for the interaction term is presented by epibenthic group in Table 3.6. When interpreted in combination with MDS ordinations, it is probable that many, if not most, of these differences were statistical rather than biological. Unique permutation values were uniformly high for both count and percent cover

macro-algal data (>6627), but varied widely for interaction pairwise comparisons for the sessile macro-invertebrate group, again suggesting that these pairwise comparisons be interpreted with caution.

The site similarity/dissimilarity matrices identified trends underlying the significant interaction term (Table 3.7). Between-site values indicated that not all spatial differences within each year follow a clear west-east progression, and the scale of inter-site similarities for some site pairs varied between years. For example, for the macro-algal community, similarity was lower between a central site (The Sirens) and eastern site (Barrett Reef) (18.36%) than between the eastern site and a western site (2nd Wash) (27.09%). Similarly, within the sessile group, the western site 2nd Wash was less similar to its neighbour, the western site Red Rocks (24.04%) and more similar with the central site Princess Bay (40.89%). Within the mobile macro-invertebrate group, 2nd Wash was more similar to the eastern site Palmer Head (48.06%) than to its nearer neighbour, The Sirens (34.67%) (Table 3.7).

3.1.3 Species contributing to spatial and interannual differences among sites: SIMPER

This routine was used to identify the percent contribution each surveyed species made to each site's assemblage structure, identified by "similarity/dissimilarity" values for each site. Because this routine can only perform pairwise tests, the larger-scale spatial factor "Area", with three levels (West, Central, East, see Figure 3.1) was used to obtain a more manageable output (Tables 3.8-3.11).

Interannual patterns

The scale of dissimilarities in assemblage structures between survey years differed among benthic groups. Interannual dissimilarities ranged from 57.13% for mobile macro-invertebrates (Table 3.10) to 94.54% for sessile macro-invertebrates (Table 3.11). The macro-algal counts subset was dissimilar at 66.66% and the percent cover set more so at 75.09% (Tables 3.8, 3.9). However, data sets were characterised by

Table 3.3. Differences in multivariate assemblage structure among baseline sites ($n = 10$) and between survey years ($n = 2$) for a two-year annual baseline survey of Wellington, New Zealand's south coast rocky-reefs. PERMANOVA results are from a two-way fixed-factor analysis of differences in location. Tests were significant at $*p < 0.05$ and these were generated from 9999 permutations. **Square roots are a measure of similarity and were calculated from estimated mean squares.

GROUP	Source	df	MS	Pseudo-F	p-value*	Unique permutations	Square root**
Macro-algae: counts (subset of 11 species)	Year	1	5620.6	6.1737	0.0002	9955	7.234
	Site	9	9436.9	10.366	0.0001	9862	21.246
	Year x Site	8	2616.8	2.8743	0.0001	9864	13.063
	Residual						30.173
cover (complete set of 48 species)	Year	1	12180.0	10.007	0.0001	9940	11.037
	Site	9	8754.9	7.1933	0.0001	9843	19.976
	Year x Site	8	3824.5	3.1423	0.0001	9849	16.147
	Residual						34.887
Sessile macro-invertebrates: percent cover (30 species)	Year	1	5305.0	3.0736	0.003	9926	6.306
	Site	9	7551.0	4.3748	0.0001	9826	17.561
	Year x Site	8	3279.8	1.9002	0.0001	9833	12.465
	Residual						41.545
Mobile macro-invertebrates: counts (36 species)	Year	1	3894.9	3.5405	0.0004	9944	8.811
	Site	9	6897.3	6.2697	0.0001	9841	27.700
	Year x Site	8	1541.1	1.4009	0.0219	9838	10.500
	Residual						33.168

Table 3.4. PERMDISP results of tests for homogeneity of dispersion in each factor used in PERMANOVA analyses. Interaction terms only are presented when significant. Samples were pre-treated as described above. Tests were significant at $p < 0.05$, with p-values generated from 9999 permutations.

GROUP	Factor	df	F	p-value*
Macro-algae (subset: 11 species): counts	Site by year	18, 171	3.9443	0.0002
Macro-algae (full survey, 48 species): percent cover	Site by year	18, 171	7.5633	0.0001
Sessile macro-invertebrates (30 species): percent cover	Site by year	18, 171	6.5839	0.0001
Mobile macro-invertebrates (36 species): counts	Year	1, 74	8.6227	0.0062
	Site	9, 66	2.1499	0.1140
	Site by year	18, 57	2.3809	0.1082

Table 3.5. Pairwise comparisons from PERMANOVA test for the significant interaction term “Year x Site”. The shorter list of non-significant results is presented here and the longer list of significant pairs is presented in Appendix G. P-values were derived from 9999 permutations and significant at $p < 0.05$.

Groups	t	p-value	Unique perm's	Groups	t	p-value	Unique perms
Sites within 2008				Sites within 2009			
Macro-algae: counts m ⁻² (subset of 11 species)							
1st Wash, Palmer Head	1.0885	0.3050	9482	1st Wash, Red Rocks	1.1740	0.2521	9156
1st Wash, Barrett Reef	1.3594	0.1410	9465	Red Rocks, Yung Pen	1.4455	0.0750	9138
The Sirens, Princess Bay	1.0983	0.3190	6661	Red Rocks, The Sirens	1.3666	0.1367	9180
The Sirens, Palmer Head	1.6952	0.0321	8728	Red Rocks, Princess Bay	1.1365	0.2978	9185
				Yung Pen, The Sirens	1.3121	0.1531	9443
				Yung Pen, Princess Bay	1.1073	0.3161	9476
				The Sirens, Princess Bay	1.0865	0.3350	9422
Macro-algae: percent cover m ⁻² (subset of 11 species)							
2nd Wash, The Sirens	1.4553	0.0653	9436	2nd Wash, Red Rocks	1.4818	0.0690	9439
Red Rocks, Yung Pen	1.3980	0.0859	9371	1st Wash, Red Rocks	0.8443	0.5792	9470
The Sirens, Princess Bay	0.9929	0.4002	9428	Red Rocks, Yung Pen	1.1019	0.3191	9418
				Red Rocks, The Sirens	1.0682	0.3547	9399
				Yung Pen, The Sirens	1.1898	0.1676	9374
				The Sirens, Princess Bay	1.1431	0.2512	9402
				Palmer Head, Barrett Reef	0.9235	0.4064	9476
				Palmer Head, Breaker Bay	1.1234	0.2826	9416
Sessile macro-invertebrates: percent cover m ⁻²							
2nd Wash, 1st Wash	1.0323	0.3697	6607	2nd Wash, 1st Wash	1.0293	0.3770	9461
2nd Wash, The Sirens	1.3542	0.0918	3208	2nd Wash, Red Rocks	1.2955	0.1318	9176
1st Wash, The Sirens	1.4257	0.0780	1859	2nd Wash, The Sirens	1.3496	0.0827	8665
1st Wash, Palmer Head	1.4764	0.0874	553	2nd Wash, Palmer Head	1.1915	0.1785	7906
Yung Pen, The Sirens	1.4323	0.0683	740	1st Wash, Palmer Head	1.3632	0.0832	6632
Yung Pen, Princess Bay	1.3800	0.0950	132	Red Rocks, The Sirens	1.4603	0.0540	6662
The Sirens, Princess Bay	1.0665	0.3041	128	Red Rocks, Palmer Head	1.4210	0.0708	4999
The Sirens, Palmer Head	1.0858	0.3206	192	Yung Pen, The Sirens	1.0537	0.3481	4962
The Sirens, Flax Bay	1.2338	0.1653	32	Yung Pen, Flax Bay	1.2765	0.1501	990
Princess Bay, Palmer Head	1.1646	0.2888	31	Yung Pen, Breaker Bay	1.1693	0.2182	1411
Princess Bay, Flax Bay	0.8745	0.8606	6	The Sirens, Princess Bay	1.1146	0.2757	7874
Palmer Head, Flax Bay	1.3015	0.1378	12	The Sirens, Palmer Head	1.3106	0.1005	3226
				The Sirens, Breaker Bay	1.3904	0.0847	1854
				Palmer Head, Flax Bay	1.3569	0.1093	504
				Palmer Head, Barrett Reef	1.2213	0.1536	7912
				Palmer Head, Breaker Bay	1.2342	0.1512	990
				Flax Bay, Breaker Bay	0.9003	0.5887	256
Mobile macro-invertebrates: counts m ⁻²							
2nd Wash, 1st Wash	1.5066	0.0566	35	2nd Wash, Palmer Head	1.3210	0.1716	35
2nd Wash, Red Rocks	1.2831	0.2303	35	1st Wash, Barrett Reef	1.7042	0.0558	35
2nd Wash, The Sirens	1.6457	0.0836	35	Red Rocks, Palmer Head	1.3394	0.0801	35
2nd Wash, Palmer Head	1.8640	0.0564	35	Yung Pen, The Sirens	1.3944	0.0592	35
1st Wash, Red Rocks	1.5392	0.0571	35	Yung Pen, Princess Bay	1.6299	0.0552	35
Red Rocks, Yung Pen	1.2728	0.2538	35	Yung Pen, Palmer Head	0.7004	0.9411	35
Red Rocks, The Sirens	1.4683	0.1088	35	The Sirens, Princess Bay	1.5802	0.0544	35
Yung Pen, The Sirens	1.4363	0.1161	35	Palmer Head, Barrett Reef	1.3198	0.1153	35
Yung Pen, Princess Bay	1.1047	0.2849	35				
The Sirens, Princess Bay	1.5220	0.0576	35				

Table 3.6. Similarities in epibenthic assemblage structure within/between baseline survey sites presented by major functional group for each level of “Year” (2007/8 and 2008/9). Similarities were defined by pairwise PERMANOVA tests on Bray-Curtis distances from percent cover macro-algal abundances and sessile macro-invertebrates) and from dispersion-weighted, square-root transformed macro-algal counts. Data were collected from Wellington, New Zealand’s south coast nearshore rocky reefs.

Macro-algae counts m⁻²: subset of 11 species										
2008										
	2nd Wash	1st Wash	Red Rocks	Yung Pen	The Sirens	Princess Bay	Palmer Head	Flax Bay	Barrett Reef	
2nd Wash	56.841									
1st Wash	51.876	67.795								
Red Rocks	46.789	42.947	64.463							
Yung Pen	42.160	42.874	52.864	54.854						
The Sirens	42.144	48.740	43.078	40.862	47.554					
Princess Bay	43.946	46.381	53.948	47.409	51.764	56.372				
Palmer Head	47.765	63.543	39.032	40.678	48.971	47.036	59.866			
Flax Bay	33.530	44.351	36.921	37.578	41.807	44.040	48.634	57.904		
Barrett Reef	55.244	64.528	43.757	42.552	43.599	43.714	58.115	39.335	64.350	
2009										
	2nd Wash	1st Wash	Red Rocks	Yung Pen	The Sirens	Princess Bay	Palmer Head	Flax Bay	Barrett Reef	Breaker Bay
2nd Wash	68.628									
1st Wash	55.905	54.764								
Red Rocks	54.659	53.071	53.221							
Yung Pen	53.681	48.974	51.312	54.621						
The Sirens	62.538	54.155	57.670	58.835	68.098					
Princess Bay	51.275	47.348	51.772	53.103	59.145	53.298				
Palmer Head	61.680	58.775	49.546	48.698	54.579	48.773	72.145			
Flax Bay	35.712	43.722	41.693	40.470	34.471	36.994	41.248	55.546		
Barrett Reef	59.018	56.844	46.133	47.085	46.693	42.324	65.981	49.442	70.654	
Breaker Bay	52.373	48.523	49.157	50.757	49.974	50.364	61.954	47.169	60.143	72.620
Macro-algae percent cover m⁻²										
2008										
	2nd Wash	1st Wash	Red Rocks	Yung Pen	The Sirens	Princess Bay	Palmer Head	Flax Bay	Barrett Reef	
2nd Wash	38.319									
1st Wash	41.076	62.432								
Red Rocks	48.710	55.296	75.818							
Yung Pen	44.661	55.219	68.203	64.478						
The Sirens	26.703	34.767	34.401	35.965	25.574					
Princess Bay	31.921	40.134	42.902	43.251	34.710	42.771				
Palmer Head	33.815	48.154	43.813	42.988	27.606	31.208	48.449			
Flax Bay	20.584	27.645	26.880	27.132	21.692	21.606	29.347	30.693		
Barrett Reef	27.093	42.501	29.253	28.438	18.357	21.058	35.065	19.493	54.302	
2009										
	2nd Wash	1st Wash	Red Rocks	Yung Pen	The Sirens	Princess Bay	Palmer Head	Flax Bay	Barrett Reef	Breaker Bay
2nd Wash	71.561									
1st Wash	55.077	55.906								
Red Rocks	58.771	56.075	53.943							
Yung Pen	61.769	53.417	56.793	60.216						
The Sirens	68.636	57.528	61.261	64.717	70.837					
Princess Bay	63.581	54.027	57.420	62.128	68.810	67.341				
Palmer Head	56.763	47.004	47.276	52.229	54.527	52.915	56.704			
Flax Bay	43.772	41.932	42.043	45.442	45.377	45.749	44.276	42.814		
Barrett Reef	62.437	54.255	53.436	57.539	60.238	58.317	62.909	53.964	75.589	
Breaker Bay	61.379	51.693	52.361	57.682	59.534	58.364	61.465	51.600	71.440	71.400

Table 3.7, continued: Mobile and sessile macro-invertebrate group similarities within/between baseline survey sites for each level of “Year” (2007/8 and 2008/9). Similarities were defined by pairwise PERMANOVA tests on Bray-Curtis distances from percent cover sessile macro-invertebrate abundances and from dispersion-weighted, square-root transformed mobile macro-invertebrate counts. Data were collected from Wellington, New Zealand’s south coast nearshore rocky reefs.

Sessile macro-invertebrates percent cover m⁻²										
2008										
	2nd Wash	1st Wash	Red Rocks	Yung Pen	The Sirens	Princess Bay	Palmer Head	Flax Bay	Barret Reef	
2nd Wash	34.751									
1st Wash	36.949	39.456								
Red Rocks	24.043	25.845	38.160							
Yung Pen	32.990	36.491	33.578	41.432						
The Sirens	37.816	41.134	26.366	42.503	49.420					
Princess Bay	40.889	44.895	29.476	51.141	57.950	66.370				
Palmer Head	42.543	49.300	26.236	47.460	57.112	65.383	65.370			
Flax Bay	43.368	48.246	28.072	53.617	63.520	74.847	71.843	82.857		
Barret Reef	21.995	25.958	18.774	21.569	22.730	22.478	26.667	23.746		
2009										
	2nd Wash	1st Wash	Red Rocks	Yung Pen	The Sirens	Princess Bay	Palmer Head	Flax Bay	Barret Reef	Breaker Bay
2nd Wash	33.149									
1st Wash	34.201	34.405								
Red Rocks	37.337	34.274	45.474							
Yung Pen	38.838	36.297	46.205	61.300						
The Sirens	35.604	31.393	41.843	52.280	44.710					
Princess Bay	38.416	33.328	44.790	57.708	49.335	55.998				
Palmer Head	34.597	34.630	40.364	49.410	41.452	43.526	42.762			
Flax Bay	39.457	38.278	49.272	65.809	54.395	56.594	56.639	73.552		
Barret Reef	26.835	27.009	30.663	34.896	28.803	31.713	33.115	36.026	31.784	
Breaker Bay	36.326	34.646	45.288	60.031	49.250	51.458	51.049	67.527	37.253	60.129
Mobile macro-invertebrates count m⁻²										
2008										
	2nd Wash	1st Wash	Red Rocks	Yung Pen	The Sirens	Princess Bay	Flax Bay	Palmer He:	Barret Reef	
2nd Wash	43.810									
1st Wash	38.418	50.094								
Red Rocks	39.156	37.099	42.714							
Yung Pen	41.216	27.020	43.353	50.590						
The Sirens	30.779	25.971	33.806	38.753	40.726					
Princess Bay	34.718	22.144	38.966	52.026	39.442	56.281				
Flax Bay	21.237	17.717	37.803	40.366	24.042	42.428	57.410			
Palmer Head	37.904	23.389	42.341	47.499	33.838	50.262	43.706	62.613		
Barret Reef	33.903	37.657	34.987	36.490	28.563	33.096	21.997	37.202	47.507	
2009										
	2nd Wash	1st Wash	Red Rocks	Yung Pen	The Sirens	Princess Bay	Flax Bay	Palmer He:	Barret Reef	Breaker Bay
2nd Wash	51.676									
1st Wash	44.164	57.607								
Red Rocks	46.877	50.268	64.155							
Yung Pen	43.269	37.483	51.241	49.632						
The Sirens	34.673	39.947	44.235	48.040	55.949					
Princess Bay	28.135	28.719	45.187	48.888	53.192	64.640				
Flax Bay	21.822	28.478	39.827	39.437	42.321	50.712	70.724			
Palmer Head	48.064	42.909	54.264	54.964	41.962	42.041	37.984	53.891		
Barret Reef	43.825	49.565	50.037	47.728	50.235	42.027	38.338	52.265	58.864	
Breaker Bay	36.362	37.229	41.915	47.622	39.150	40.727	40.252	49.356	46.248	59.465

Table 3.7. Macro-algal species contribution to dissimilarity considered for each level of “Year” and “Area” using count data for an 11-species subset of the full macro-algal species survey list. SIMPER used dispersion-weighted, square-root transformed data with a 90% contribution cut-off point. Data were collected from Wellington’s south coast shallow rocky-reef communities during two annual surveys in 2007/08 and 2008/09.

Variable	Species	Average Abundance	Average dissimilarity /SD	Dissimilarity /SD	Percent contribution	Cumulative contribution
YEAR						
Groups 2008 & 2009: Average dissimilarity = 66.66						
2008/2009						
	<i>Ecklonia radiata</i>	0.73/1.04	14.43	0.97	21.64	21.64
	<i>Landsburgia quercifolia</i>	0.41/0.53	10.21	0.95	15.32	36.96
	<i>Lessonia variegata</i>	0.52/0.57	9.92	0.74	14.87	51.83
	<i>Carpophyllum maschalocarpum</i>	0.31/0.38	7.92	0.89	11.88	63.72
	<i>Carpophyllum flexuosum</i>	0.12/0.37	6.83	0.66	10.24	73.96
	<i>Marginariella</i> spp.	0.15/0.15	4.07	0.45	6.10	80.06
	<i>Glossophora kunthii</i>	0.11/0.20	3.67	0.53	5.50	85.56
	<i>Macrocystis pyrifera</i>	0.16/0.08	3.34	0.43	5.01	90.57
AREA						
Groups West & Central: Average dissimilarity = 69.81						
	<i>Lessonia variegata</i>	0.79/0.83	14.24	0.95	20.40	20.40
	<i>Ecklonia radiata</i>	0.90/0.48	13.54	0.99	19.40	39.80
	<i>Landsburgia quercifolia</i>	0.49/0.60	11.31	0.95	16.20	56.00
	<i>Carpophyllum maschalocarpum</i>	0.24/0.39	7.83	0.86	11.22	67.22
	<i>Marginariella</i> spp.	0.28/0.10	5.63	0.47	8.06	75.28
	<i>Carpophyllum flexuosum</i>	0.19/0.23	5.41	0.61	7.75	83.03
	<i>Glossophora kunthii</i>	0.01/0.28	4.16	0.61	5.96	88.99
	<i>Sargassum sinclairii</i>	0.06/0.22	4.07	0.49	5.82	94.81
Groups West & East: Average dissimilarity = 70.16						
	<i>Ecklonia radiata</i>	0.90/1.25	16.51	1.04	23.53	23.53
	<i>Lessonia variegata</i>	0.79/0.10	12.91	0.96	18.39	41.93
	<i>Landsburgia quercifolia</i>	0.49/0.35	9.33	0.92	13.30	55.23
	<i>Carpophyllum maschalocarpum</i>	0.24/0.41	8.06	0.85	11.48	66.71
	<i>Carpophyllum flexuosum</i>	0.19/0.32	6.9	0.66	9.84	76.55
	<i>Macrocystis pyrifera</i>	0.00/0.31	5.25	0.56	7.49	84.03
	<i>Marginariella</i> spp.	0.28/0.07	5.01	0.48	7.15	91.18
Groups Central & East: Average dissimilarity = 75.40						
	<i>Ecklonia radiata</i>	0.48/1.25	16.2	1.11	21.49	21.49
	<i>Lessonia variegata</i>	0.83/0.10	12.77	1.18	16.94	38.43
	<i>Landsburgia quercifolia</i>	0.60/0.35	9.81	0.99	13.01	51.44
	<i>Carpophyllum maschalocarpum</i>	0.39/0.41	8.02	0.94	10.64	62.08
	<i>Carpophyllum flexuosum</i>	0.23/0.32	6.24	0.68	8.28	70.36
	<i>Glossophora kunthii</i>	0.28/0.17	5.48	0.66	7.27	77.63
	<i>Macrocystis pyrifera</i>	0.00/0.31	5.42	0.51	7.19	84.82
	<i>Sargassum sinclairii</i>	0.22/0.08	4.24	0.45	5.62	90.44

Table 3.8. Macro-algal contribution to dissimilarity for each level of “Year” and “Area” using **percent cover** data for all species observed. SIMPER used a 90% contribution cut-off point. Data were collected from Wellington's south coast shallow rocky-reef communities during two annual surveys in 2007/08 and 2008/09.

Variable Species/Group	Average Abundance	Average dissimilarity	Dissimilarity /SD	Percent contribution	Cumulative contribution
YEAR					
Groups 2008 & 2009: Average dissimilarity = 75.09					
	2008/2009				
<i>Ecklonia radiata</i>	16.51/21.02	14.83	0.92	19.75	19.75
<i>Lessonia variegata</i>	14.20/15.90	12.11	0.78	16.12	35.87
<i>Caulerpa brownii</i>	13.08/7.23	9.98	0.60	13.29	49.16
<i>Carpophyllum maschalocarpum</i>	5.02/3.31	4.79	0.59	6.38	55.54
<i>Carpophyllum flexuosum</i>	1.49/4.06	3.63	0.48	4.83	60.37
Geniculate coralline algae	4.28/3.89	3.57	0.82	4.76	65.13
<i>Zonaria turneriana</i>	3.23/2.29	3.09	0.72	4.11	69.24
<i>Marginariella</i> spp	4.21/1.43	2.76	0.29	3.68	72.92
<i>Halopteris</i> sp.	2.61/0.90	2.01	0.52	2.68	75.60
<i>Landsbergia quercifolia</i>	1.64/1.97	1.88	0.69	2.51	78.11
Turfing foliose red (unknown)	1.80/0.67	1.55	0.51	2.06	80.17
AREA					
Groups West & Central: Average dissimilarity = 77.20					
	2008/2009				
<i>Lessonia variegata</i>	24.38/21.78	18.63	1.14	24.13	24.13
<i>Caulerpa brownii</i>	16.78/14.88	16.21	0.85	21.00	45.12
<i>Ecklonia radiata</i>	15.15/4.82	9.35	0.83	12.11	57.24
<i>Marginariella</i> spp.	7.36/0.64	4.47	0.35	5.79	63.03
Geniculate coralline algae	2.72/6.37	4.47	0.94	5.79	68.82
<i>Carpophyllum maschalocarpum</i>	2.92/3.20	3.26	0.55	4.22	73.04
<i>Halopteris</i> sp.	1.35/3.73	2.91	0.67	3.77	76.80
<i>Carpophyllum flexuosum</i>	1.75/2.92	2.84	0.42	3.68	80.48
Groups West & East: Average dissimilarity = 83.11					
	2008/2009				
<i>Ecklonia radiata</i>	15.15/34.14	19.88	1.17	23.92	23.92
<i>Lessonia variegata</i>	24.38/1.40	15.58	0.91	18.75	42.66
<i>Caulerpa brownii</i>	16.78/0.00	11.31	0.62	13.61	56.28
<i>Carpophyllum maschalocarpum</i>	2.92/5.94	4.83	0.61	5.81	62.09
<i>Marginariella</i> spp	7.36/0.60	4.05	0.34	4.87	66.96
<i>Carpophyllum flexuosum</i>	1.75/3.71	3.45	0.51	4.15	71.11
<i>Zonaria turneriana</i>	2.43/3.33	2.92	0.71	3.52	74.63
Geniculate coralline algae	2.72/3.27	2.39	1.06	2.87	77.50
<i>Euptilota formisissima</i>	2.57/1.06	2.05	0.82	2.47	79.97
<i>Macrocystis pyrifera</i>	0.00/2.86	1.79	0.35	2.15	82.13
Groups Central & East: Average dissimilarity = 86.20					
	2008/2009				
<i>Ecklonia radiata</i>	4.82/34.14	21.29	1.14	24.70	24.70
<i>Lessonia variegata</i>	21.78/1.40	15.00	0.91	17.40	42.10
<i>Caulerpa brownii</i>	14.88/0.00	8.57	0.57	9.94	52.04
<i>Carpophyllum maschalocarpum</i>	3.20/5.94	5.47	0.61	6.35	58.39
Geniculate coralline algae	6.37/3.27	4.72	0.94	5.48	63.87
<i>Carpophyllum flexuosum</i>	2.92/3.71	4.40	0.50	5.11	68.97
<i>Zonaria turneriana</i>	2.34/3.33	3.14	0.74	3.64	72.62
<i>Halopteris</i> sp.	3.73/0.29	2.58	0.62	2.99	75.60
<i>Macrocystis pyrifera</i>	0.00/2.86	1.97	0.35	2.29	77.89
<i>Landsbergia quercifolia</i>	2.18/1.09	1.93	0.69	2.24	80.13

Table 3.9. Mobile macro-invertebrate contribution to dissimilarity for each level of “Year” and “Area” from **count** data. SIMPER used dispersion-weighted, square-root transformed data with a 70% contribution cut-off point. Data were collected from Wellington's south coast shallow rocky-reef communities in 2008-09.

Variable Species/Group	Average Abundance	Average dissimilarity	Dissimilarity /SD	Percent contribution	Cumulative contribution
YEAR					
Groups 2008 & 2009: Average dissimilarity = 57.13					
2008/2009					
<i>Pentagonaster pulchellus</i>	0.96/1.44	4.37	1.26	7.65	7.65
<i>Patiriella regularis</i>	1.39/1.54	4.07	1.37	7.12	14.77
<i>Astrostele scabra</i>	0.66/1.31	3.64	1.28	6.37	21.14
<i>Diplodontias dilatatus</i>	0.47/0.88	3.60	1.16	6.30	27.44
<i>Evechinus chloroticus</i>	0.85/1.23	3.47	1.09	6.08	33.52
<i>Australostichopus mollis</i>	0.72/1.01	3.24	1.24	5.67	39.19
<i>Haliotis australis</i>	0.69/1.00	2.81	1.16	4.92	44.11
<i>Turbo smaragdus</i>	0.80/0.73	2.61	1.07	4.57	48.67
<i>Plagusia chabris</i>	0.42/0.62	2.41	1.08	4.22	52.89
<i>Haliotis iris</i>	0.61/0.79	2.31	1.05	4.05	56.94
<i>Jasus edwardsii</i>	0.28/0.48	2.06	0.90	3.60	60.54
<i>Pectinura maculata</i>	0.42/0.60	2.04	0.88	3.57	64.12
<i>Cookia sulcata</i>	0.40/0.36	1.98	0.94	3.47	67.59
<i>Aphelodoris luctosa</i>	0.15/0.50	1.89	0.92	3.30	70.89
AREA					
Groups West & Central: Average dissimilarity = 63.84					
2008/2009					
<i>Turbo smaragdus</i>	0.13/1.33	5.45	1.48	8.53	8.53
<i>Haliotis australis</i>	0.34/1.52	5.30	1.66	8.29	16.82
<i>Patiriella regularis</i>	1.08/1.49	4.37	1.24	6.84	23.66
<i>Astrostele scabra</i>	0.67/1.32	4.32	1.31	6.77	30.43
<i>Haliotis iris</i>	0.07/1.03	4.14	1.54	6.48	36.91
<i>Pentagonaster pulchellus</i>	1.08/0.95	4.09	1.28	6.41	43.31
<i>Diplodontias dilatatus</i>	0.81/0.55	3.71	1.05	5.81	49.12
<i>Evechinus chloroticus</i>	1.33/0.61	3.45	1.15	5.41	54.53
<i>Plagusia chabris</i>	0.43/0.80	3.12	1.28	4.89	59.42
<i>Australostichopus mollis</i>	0.79/0.42	3.01	1.21	4.72	64.14
<i>Jasus edwardsii</i>	0.44/0.46	2.50	0.94	3.91	68.05
<i>Cookia sulcata</i>	0.43/0.44	2.48	1.02	3.88	71.94
Groups West & East: Average dissimilarity = 62.80					
2008/2009					
<i>Patiriella regularis</i>	1.08/1.78	4.82	1.37	7.68	7.68
<i>Pentagonaster pulchellus</i>	1.08/1.55	4.41	1.16	7.02	14.70
<i>Pectinura maculata</i>	0.14/1.12	4.12	1.20	6.57	21.27
<i>Australostichopus mollis</i>	0.79/1.33	4.08	1.36	6.49	27.76
<i>Evechinus chloroticus</i>	1.33/1.19	3.71	1.23	5.91	33.66
<i>Diplodontias dilatatus</i>	0.81/0.69	3.69	1.18	5.87	39.53
<i>Haliotis iris</i>	0.07/0.95	3.59	1.18	5.72	45.26
<i>Astrostele scabra</i>	0.67/1.01	3.12	1.17	4.96	50.22
<i>Turbo smaragdus</i>	0.13/0.81	3.08	0.91	4.90	55.12
<i>Haliotis australis</i>	0.34/0.72	2.61	1.24	4.15	59.27
<i>Scutus breviculus</i>	0.03/0.71	2.51	0.78	4.00	63.27
<i>Plagusia chabris</i>	0.43/0.36	2.14	0.92	3.41	66.68
<i>Cookia sulcata</i>	0.43/0.29	1.93	0.96	3.08	69.76
<i>Jasus edwardsii</i>	0.44/0.26	1.79	0.91	2.85	72.61

Table 3.10, page 2. Mobile macro-invertebrate contribution to dissimilarity among Wellington south coast's nearshore rocky-reef assemblages.

Variable Species/Group	Average Abundance	Average dissimilarity	Dissimilarity /SD	Percent contribution	Cumulative contribution
Groups Central & East: Average dissimilarity = 58.31					
2008/2009					
<i>Patiriella regularis</i>	1.49/1.78	3.96	1.35	6.79	6.79
<i>Pentagonaster pulchellus</i>	0.95/1.55	3.91	1.23	6.71	13.49
<i>Turbo smaragdus</i>	1.33/0.81	3.76	1.51	6.46	19.95
<i>Australostichopus mollis</i>	0.42/1.33	3.74	1.29	6.41	26.36
<i>Haliotis australis</i>	1.52/0.72	3.55	1.40	6.09	32.45
<i>Pectinura maculata</i>	0.20/1.12	3.44	1.21	5.90	38.35
<i>Evechinus chloroticus</i>	0.61/1.19	3.06	0.99	5.25	43.60
<i>Haliotis iris</i>	1.03/0.95	2.94	1.37	5.04	48.63
<i>Astrostele scabra</i>	1.32/1.01	2.76	1.10	4.73	53.36
<i>Scutus breviculus</i>	0.33/0.71	2.44	1.02	4.18	57.54
<i>Plagusia chabris</i>	0.80/0.36	2.31	1.27	3.97	61.51
<i>Diplondontias dilatatus</i>	0.55/0.69	2.26	0.94	3.87	65.38
<i>Cookia sulcata</i>	0.44/0.29	1.69	0.90	2.90	68.28
<i>Aphelodoris luctosa</i>	0.39/0.40	1.66	0.95	2.84	71.12

Table 3.10. Sessile macro-invertebrate contribution to dissimilarity between levels of “Year” and “Area”. SIMPER used percent cover data with an 80% contribution cut-off point. Data were collected from Wellington, New Zealand’s south coast shallow rocky-reef communities in summers 2007/8 and 2008/9.

Variable Species/Group	Average Abundance	Average dissimilarity	Dissimilarity /SD	Percent contribution	Cumulative contribution
YEAR					
Groups 2008 & 2009: Average dissimilarity = 94.54					
2008/2009					
Bryozoa G	0.23/0.14	9.86	0.52	10.43	10.43
Ascidia D	0.14/0.09	8.18	0.45	8.66	19.09
Porifera B	0.12/0.11	8.09	0.47	8.55	27.64
Hydrozoa B	0.14/0.08	6.88	0.48	7.28	34.92
Porifera E	0.07/0.09	6.02	0.39	6.37	41.29
Ascidia A	0.05/0.10	5.17	0.33	5.47	46.76
Hydrozoa A	0.10/0.14	4.77	0.42	5.04	51.80
Porifera <i>Tethya</i> sp.	0.07/0.07	4.45	0.34	4.70	56.51
Bryozoa B	0.03/0.08	4.14	0.29	4.38	60.89
Porifera F	0.07/0.14	3.89	0.25	4.12	65.01
Porifera L	0.12/0.02	3.78	0.30	3.99	69.00
Bryozoa E	0.08/0.03	3.21	0.27	3.39	72.39
AREA					
Groups West & Central: Average dissimilarity = 94.43					
2008/2009					
Bryozoa G	0.36/0.04	10.74	0.62	11.37	11.37
Porifera B	0.16/0.19	9.77	0.63	10.35	21.72
Hydrozoa B	0.22/0.09	9.42	0.62	9.98	31.70
Hydrozoa A	0.27/0.00	9.15	0.56	9.69	41.39
Ascidia D	0.16/0.07	6.63	0.59	7.02	48.41
Porifera E	0.10/0.08	6.16	0.45	6.52	54.93
Porifera <i>Tethya</i> sp.	0.16/0.04	5.92	0.48	6.27	61.20
Ascidia A	0.06/0.07	4.23	0.33	4.47	65.68
Porifera F	0.03/0.13	3.85	0.27	4.08	69.75
Porifera I	0.11/0.02	3.74	0.41	3.96	73.71
Groups West & East: Average dissimilarity = 95.46					
2008/2009					
Bryozoa G	0.36/0.16	12.52	0.66	13.12	13.12
Hydrozoa A	0.27/0.09	10.18	0.59	10.67	23.78
Ascidia D	0.16/0.10	7.01	0.51	7.34	31.13
Hydrozoa B	0.22/0.03	6.95	0.51	7.28	38.41
Ascidia A	0.06/0.09	5.07	0.38	5.31	43.72
Porifera <i>Tethya</i> sp.	0.16/0.02	5.05	0.45	5.29	49.01
Porifera E	0.10/0.06	4.73	0.42	4.96	53.96
Bryozoa B	0.00/0.14	4.10	0.35	4.30	58.26
Porifera B	0.16/0.01	4.03	0.41	4.22	62.49
Porifera F	0.03/0.14	3.48	0.25	3.65	66.13
Porifera I	0.11/0.00	3.18	0.38	3.33	69.46
Porifera D	0.08/0.10	3.18	0.38	3.33	72.79
Groups Central & East: Average dissimilarity = 97.86					
2008/2009					
Porifera B	0.19/0.01	12.05	0.58	12.31	12.31
Ascidia D	0.07/0.10	8.49	0.44	8.67	20.98
Bryozoa G	0.04/0.16	7.83	0.41	8.00	28.98
Porifera E	0.08/0.06	7.42	0.41	7.59	36.56
Hydrozoa B	0.09/0.03	6.72	0.44	6.87	43.43
Bryozoa B	0.00/0.14	6.30	0.40	6.44	49.87
Ascidia A	0.07/0.09	6.00	0.36	6.14	56.00
Porifera F	0.13/0.14	5.21	0.29	5.33	61.33
Porifera <i>Tethya</i> sp.	0.04/0.02	4.09	0.28	4.18	65.51
Porifera L	0.04/0.06	3.90	0.29	3.98	69.49
Ascidia C	0.00/0.17	3.60	0.27	3.68	73.17

similar proportions of species: 23% of the macro-algal species and 38% and 33% for the mobile and sessile macro-invertebrates respectively.

The number of species and functional groups needed to characterise macro-algal count versus percent cover data differed. Percentage values that follow are the contributions to dissimilarity (calculated to the nearest 0.01% in the analyses) and not mean values of observed cover (calculated to the nearest 1%). Patterns in the 11-species macro-algal counts data set were explained by five canopy species contributing a total of 74.96% to dissimilarity between years: *E. radiata*, *L. quercifolia*, *L. variegata*, *C. flexuosum* and *C. maschalocarpum* (Table 3.8). Two additional canopy and one understorey species contributed the remaining 16.61% dissimilarity. The percent cover set required nine species, although six of these were shared with the count list, above. Two functional groups, two understorey and two turfing, contributed to patterns in percent cover data. The giant kelp *Macrocystis pyrifera* contributed to interannual differences in assemblage patterns (5.01%) (Table 3.8).

Individual mobile macro-invertebrate species contributed far less than macro-algal species to interannual dissimilarities: less than 10% (Table 3.9). Fourteen species defined the differences in assemblage structure between years. Of the top five contributors, only one was a species of recreational or commercial interest, the kina *Evechinus chloroticus*. The other four highest-contributing species were Asteroids (sea stars).

Among sessile macro-invertebrates only Bryozoa G contributed more than 10% (Table 3.11). The top five contributors included all observed sessile taxa. The remaining contributors were of mixed taxonomic origin.

Spatial patterns

Differences in assemblage composition among areas for the macro-algal count data were defined by two functional phaeophyte groups. Most of the dissimilarities among the areas "west", "central", and "east" (62.08-67.22%) were defined by four Phaeophyta canopy species contributing more than 10% each: *Ecklonia radiata*, *Lessonia variegata*, *Carpophyllum maschalocarpum*, and *Landsburgia quercifolia* (Table 3.8). Other species contributing less than 10% included *C. flexuosum* to all areas. At the western and central areas *Marginariella* spp. contributed 8.06%. At the eastern end the giant kelp *M. pyrifera* contributed 7.19 and 7.49%. The canopy species

Sargassum sinclairii and the only understorey species *Glossophora kunthii* contributed to the central area. Dissimilarities among areas for count data were lower (69.81-75.40%) than those identified for percent cover data (77.20-86.20%) (Tables 3.8 and 3.9).

Similarities between areas for percent cover data included a third functional group and a member of the Rhodophyta, in addition to phaeophytes (Table 3.9). Three species contributed more than 10% each to all three among-area comparisons: the canopy species *Lessonia variegata* and *Ecklonia radiata*, and the understorey *Caulerpa* spp. The turfing group “geniculate coralline algae” and the understorey species *Halopteris* spp., *Zonaria turneriana*, and the rhodophyte *Euptilota formississima* comprised the remaining percentages. Dissimilarities between area pairs were lowest for “West vs. Central” (77.20%) and highest for “Central vs. East” (86.20%). Only 17-21% of the species surveyed contributed to spatial dissimilarities for this group.

Similarity of assemblages was higher for the mobile macro-invertebrate group than for macro-algal groups (Table 3.10). Individual mobile macro-invertebrates again contributed less than 10% to dissimilarities between areas. Species of commercial or recreational importance here also ranked low in the top five list: the paua *Halotis iris* for “West vs. Central” (6.48%) and kina for “West vs. East” (5.91%). The sea star *Patiriella regularis* featured in the five top contributors to all area comparisons. Sea stars, a top shell, a sea cucumber, and the paua *H. australis* completed the top five list for all areas. Dissimilarities between area pairs were similar for “West vs Central” (63.84%) and “West vs. East” (62.80%), but “Central vs. East” was more similar (58.31%). Thirty-three to thirty-nine percent of all species surveyed contributed to spatial dissimilarities for this group.

Sessile macro-invertebrate assemblage structures were the most dissimilar of the three epibenthic groups, with a minimum value of 94.43% identified for the “West” vs. “Central” areas (Table 3.11). Three taxa featured in the top five species contributing to dissimilarities among all areas: Ascidia A, Bryozoa G, and Hydrozoa B (Table 3.11). At least one taxon contributed more than 10% to differences between areas. Dissimilarities among area pairs were highest for this group, ranging from 94.43% for West vs. Central to 97.86% for Central vs. East. Twenty-eight to thirty-three percent of all species surveyed contributed to spatial dissimilarities for this group.

3.1.4 Patterns in size-frequency distributions of dominant macro-algal species and recreationally important mobile macro-invertebrate species

Here I described and quantified spatial and temporal trends in sizes of dominant macro-algal species and mobile macro-invertebrate species of recreational interest.

Macro-invertebrate size-frequency classes showed spatial variation in both survey years (Figure 3.7). Although mean kina test diameters at all sites were ~100mm, size classes were patchily distributed at WSC's western and central sites with the exception of the western site, 1st Wash. At this site, a normal distribution was observed in both years, with higher abundances in the second survey (2009). A normal distribution of size classes was observed at all eastern sites except for Flax Bay during the first survey (2008). New recruits (<20mm) were not observed at any sites.

Patterns in size-class patterns for both paua species were characterised by discontinuous distributions of classes at the western sites, and a wider size-range observed from the central Princess Bay site eastward. When observed at the western sites, the shell lengths of the lone individuals of both species present corresponded to legal size (*H. iris*, 125mm; *H. australis*, 80mm). More size classes for both species were observed during the second survey, when greater abundances of both species were also noted. For both species, the range size of classes was greatest at central Princess Bay, and at eastern Flax Bay site for blackfoot paua. Smaller, newly-recruited size classes were not observed at any site for either species.

Distribution of size-classes varied among macro-algal canopy species, with species-specific patterns among sites and between years. One canopy species, *Ecklonia radiata*, was recorded at all baseline sites in both years. Differences in patterns between the two annual surveys might have been due to higher observed abundances noted for all species except *Macrocystis pyrifera*. Plant lengths for the two *Carpophyllum* species were positively skewed and dominated by smaller classes at all sites for both years. In contrast, *Ecklonia radiata* size classes were more normally distributed at all sites, but only during the second annual survey were the smallest size classes noted at western and central sites. *Landsburgia quercifolia* sizes were more evenly distributed among classes, with the smallest size class present at most sites in

both years. Larger individuals were present at the western and two central sites in both surveys.

Of all species, patterns in *Lessonia variegata* size classes were the most varied. In the first survey, size classes were discontinuous but marked by larger plants when present. Size classes were more normally distributed during the second survey. In both years, observed abundances were lowest at the eastern sites. The size distribution of *Macrocytis pyrifera*, present only at one eastern site, was positively skewed with only a few large individuals observed and this pattern was consistent across both years. *Marginariella boryana* was only assessed during the second survey and was only found at five sites. Sizes were normally distributed only at the westernmost site (2nd Wash), with smaller classes dominating the remaining sites.

Figure 3.7. Mobile macro-invertebrates: *Evechinus chloroticus* (kina) size-frequency distributions by site and by year from west (top) to east, and pooled across sites (bottom). No kina were observed at Red Rocks in 2008 or at 2nd Wash in 2009.

Figure 3.7, continued. Mobile macro-invertebrates: *Haliotis australis* (yellowfoot paua). No yellowfoot paua were observed at Red Rocks, Yung Pen, Palmer Head, Flax Bay, or Barrett Reef in 2008. None were observed at 2nd Wash, 1st Wash, or Palmer Head in 2009.

Figure 3.7, continued. Mobile macro-invertebrates: *Haliotis iris* (Blackfoot paua). No blackfoot paua were observed at 1st Wash, Red Rocks, or Barret Reef in 2008. None were observed at 2nd Wash, 1st Wash, or Red Rocks in 2009.

Figure 3.7, continued. Macro-algae: *Carpophyllum flexuosum*. None were observed at 2nd Wash in 2008 or 1st Wash in 2009.

Figure 3.7, continued. Macro-algae: *Carpophyllum maschalocarpum*. None were observed at 2nd Wash in 2008 and 2009 or at 1st Wash in 2009.

Figure 3.7, continued. Macro-algae: *Ecklonia radiata*. Observed at all sites in both years. *Breaker Bay was only surveyed during the second year.

Figure 3.7, continued. Macro-algae: *Landsburgia quercifolia*. None observed at Flax Bay in either survey

Figure 3.7, continued. Macro-algae: *Lessonia variegata*. No plants were observed at Palmer Head, Flax Bay, or Breaker Bay in 2008 or at Flax Bay or Barrett Reef in 2009.

Figure 3.7, final. Macro-algae: *Macrocystis pyrifera* and *Marginariella boryana*. *M. pyrifera* was only observed at eastern baseline sites, so data from these two sites were pooled for summary and presentation. *M. boryana* is presented for 2009 only due to misidentification during the previous survey.

3.2 Aim 2: Best practices for future monitoring

I addressed this aim by 1) testing for differences in abundance estimates derived from macro-algal counts versus percent cover metrics and 2) identifying representative “indicator” subsets from full species lists for expedited post-reserve monitoring of WSC rocky-reef epibenthic community structure.

3.2.1 *Abundance metrics for monitoring macro-algae: counts versus percent cover using the RELATE routine*

The RELATE routine that was used to test the null hypothesis of no match between assemblage structures based on counts versus percent cover was significant ($\rho = 0.791$, $p = 0.001$). Therefore, the use of counts as a measure of abundance will provide multivariate patterns that differ significantly from those described by percent cover data. The use of these two metrics also led to different dissimilarity values and functional groups identified by SIMPER (see above). Therefore the choice of abundance measures can produce different perceptions of assemblage patterns and species driving these patterns.

3.2.2 *Identification of structural redundancies and representative species for future monitoring: BESTStep*

This routine was used to identify species or groups of species that modelled the multivariate patterns of the full data set. At least two subsets per epibenthic group were identified as representative of full data sets (all $p < 0.001$) (Table 3.12), allowing rejection of the null hypothesis of no significant relationship between full and subset biotic matrices. Because rho values for the most parsimonious subsets differed only at the third decimal place, two subsets per epibenthic group are presented here for comparison. The composition of both subsets for each group differed by no more than three species.

The macro-algal counts group (original $n=11$) was characterised by two subsets composed only of canopy species, one with five ($\rho = 0.953$) and the second with eight ($\rho = 0.956$) (Table 3.12). The second group differed by the inclusion of *Marginariella* spp., *Sargassum sinclairii*, and *Undaria pinnatifida*. The macro-algal percent cover group containing “crustose coralline algae” (original $n=48$ species/group) identified five canopy species in each of the two subsets ($\rho = 0.961$, $\rho = 0.957$ respectively). The

difference was the inclusion of *Marginariella* spp. in the first model and *Carpophyllum maschalocarpum* in the second. Once CCA were excluded from analyses, understorey species appeared in both subsets: *Zonaria turneriana* ($p = 0.953$) and *Halopteris* spp. ($p = 0.950$).

The mobile macro-invertebrate data set (original $n = 36$) required larger subsets to represent the full assemblage patterns: 11 species for each subset identified as significant representatives of the full model ($p = 0.958$, $p = 0.952$ respectively) (Table 3.12). The first group differed by the inclusion of *Plagusia chabris* (red rock crab), while the second group differed with *Scutus breviculus* (duckbill limpet) and *Cantharidus purpureus* (top shell). The sessile macro-invertebrate group (original $n=30$) required the largest subsets of all, with 13 species in each ($p = 0.956$, $p = 0.951$ respectively) (Table 3.12). The two subsets differed by only one poriferan species (Type G versus Type F). Poriferans dominated these subsets, reflecting their dominance of this functional group in the field.

Table 3.11. Species subsets identified by BEST BVStep analyses as representative of the full species survey list for each epibenthic group. Data were collected annually for baseline surveys of Wellington, New Zealand's shallow rocky-reef communities (2007/08 and 2008/09). * indicate species unique to each group. P was significant at < 0.05.

Group	Subset A	rho p-value	Subset B	rho p-value
Macro-algae				
<i>Counts: Original set of 11 species</i>				
	<i>Carpophyllum flexuosum</i>	0.953	<i>Carpophyllum flexuosum</i>	0.956
	<i>Carpophyllum maschalocarpui</i>	p< 0.001	<i>Carpophyllum maschalocarpui</i>	p< 0.001
	<i>Ecklonia radiata</i>		<i>Ecklonia radiata</i>	
	<i>Lessonia variegata</i>		<i>Lessonia variegata</i>	
	<i>Landsburgia quercifolia</i>		<i>Landsburgia quercifolia</i>	
			<i>Marginariella</i> spp*	
			<i>Sargassum sinclairii</i> *	
			<i>Undaria pinnatifida</i> *	
<i>Percent cover: all species/genera surveyed (48)</i>				
	<i>Caulerpa</i> spp	0.961	<i>Carpophyllum maschalocarpui</i>	0.957
	<i>Crustose coralline algae</i>	p< 0.001	<i>Caulerpa</i> spp	p< 0.001
	<i>Ecklonia radiata</i>		<i>Crustose coralline algae</i>	
	<i>Lessonia variegata</i>		<i>Ecklonia radiata</i>	
	<i>Marginariella</i> spp		<i>Lessonia variegata</i>	
<i>Percent cover: crustose coralline algae omitted</i>				
	<i>Carpophyllum flexuosum</i>	0.953	<i>Carpophyllum flexuosum</i>	0.950
	<i>Carpophyllum maschalocarpui</i>	p< 0.001	<i>Carpophyllum maschalocarpui</i>	p< 0.001
	<i>Caulerpa</i> spp		<i>Caulerpa</i> spp	
	Geniculate coralline algae		Geniculate coralline algae	
	<i>Ecklonia radiata</i>		<i>Ecklonia radiata</i>	
	<i>Lessonia variegata</i>		<i>Halopteris</i> sp*	
	<i>Zonaria turneriana</i> *		<i>Lessonia variegata</i>	
Mobile macro-invertebrates: 36 species/genera surveyed				
	<i>Haliotis australis</i>	0.958	<i>Haliotis australis</i>	0.952
	<i>Haliotis iris</i>	p< 0.001	<i>Haliotis iris</i>	p< 0.001
	<i>Evechinus chloroticus</i>		<i>Evechinus chloroticus</i>	
	<i>Jasus edwardsii</i>		<i>Jasus edwardsii</i>	
	<i>Plagusia chabris</i> *		<i>Scutus breviculus</i> *	
	<i>Stichopus mollis</i>		<i>Stichopus mollis</i>	
	<i>Patiriella regularis</i>		<i>Patiriella regularis</i>	
	<i>Astrostele scabrus</i>		<i>Astrostele scabrus</i>	
	<i>Pentagonaster pulchellus</i>		<i>Pentagonaster pulchellus</i>	
	<i>Pectinura maculata</i>		<i>Pectinura maculata</i>	
	<i>Turbo smaragdus</i>		<i>Cantharidus purpureus</i> *	
Sessile macro-invertebrates: 30 species/genera surveyed				
	Bryozoan_A	0.956	Bryozoan_A	0.951
	Hydrozoan A	p< 0.001	Hydrozoan A	p< 0.001
	Hydrozoan B		Hydrozoan B	
	Poriferan A		Poriferan A	
	Poriferan B		Poriferan B	
	Poriferan E		Poriferan E	
	Poriferan G		Poriferan F*	
	Poriferan <i>Tethya</i> sp		Poriferan <i>Tethya</i> sp	
	Poriferan M		Poriferan M	
	Poriferan O		Poriferan O	
	Ascidian A		Ascidian A	
	Ascidian B		Ascidian B	
	Ascidian D		Ascidian D	

3.3 Aim 3: Describe and quantify interactions between environmental variables and epibenthic assemblage structures

I addressed this aim using three methods. First, I identified patterns in the environmental data set using non-metric multi-dimensional scaling. Second, I tested for significant relationships between environmental factors and each of the epibenthic assemblages (distance-based linear modelling). I visually presented the results of these tests using vector plots of significant factors overlaid on constrained principal components ordinations of the three epibenthic groups' biological structures (distance-based redundancy analyses). Finally, I tested for differences in total species number and in density by seabed class.

3.3.1 General patterns in environmental variables: MDS

Spatial differences among sites, identified by distances between site symbols in the MDS plots (Figures 3.8A, B), were greatest between the western sites and central and eastern sites when all environmental factors were included. Central and eastern sites also differed, although the distances were smaller and less distinctly separated. Conversely, these two patterns disappeared in an MDS plot on a reduced data set from which the variables "Distance from harbour", "Urban development", and "Hydrology (Karori Rip yes/no)" had been removed. These three were removed because all were defined by spatial gradients that possibly dominated, so structured, the spatial patterns that I was attempting to identify. In a sense they contributed to a circular definition. A very different pattern emerged after their removal: although eastern sites separated from central and western sites, areas separated less cleanly. Flax Bay remained the most isolated point but this was more pronounced in the reduced MDS (Figure 3.8B).

Lack of sediment contributed to central and some western sites, and sediment presence to the eastern sites. Mobile habitat ("Boulders", "Sand/Gravel") also separated the eastern sites and especially Flax Bay. The remaining central and eastern sites showed no clear separation or gradient, with the westernmost site "2nd Wash" placed near central sites. "Vertical reef" and "fetch" contributed to western sites and the central site "The Sirens". Interannual variation in environmental values was indicated by distances between first and second year survey sites in both plots.

Without the influence of the spatially-defined variables, the greatest interannual difference in environmental values occurred at the eastern Flax Bay site; this was due to physical habitat.

The water quality parameters “salinity”, “temperature”, and “dissolved oxygen” were strongly correlated with sites, as indicated by vector length. These correlations were paradoxical given the very narrow ranges in values identified for these three variables in Chapter 2 and I can offer no explanation for their inclusion and the scale of their contribution (vector length).

Figure 3.8A. Spatio-temporal patterns in environmental variables along Wellington's south coast using all assessed variables. The relationships and correlations (Spearman) between the variables and baseline survey sites are indicated by the angles and lengths of the vector plot: fetch, Multi-dimensional ordinations used normalised data in a Euclidean-distance matrix.

Figure 3.8B. Spatio-temporal patterns in environmental variables along Wellington's south coast defined by a reduced data set using multi-dimensional scaling on normalised data in Euclidean space. Here the variables "distance from harbour", "hydrology", and "urban development" were deleted because of their possible role as confounding variables in structuring spatial patterns.

3.3.2 Relationships between biotic and abiotic matrices: DistLM and dbRDA

Distance-based linear models identified the relative correlations of environmental parameters with each group's biotic structure. These patterns were then visualised in distance based redundancy plots (Figures 3.9-3.12). Variable correlations differed between models (Table 3.13). Moreover, although the models explained over half of the fitted variability in the biotic assemblages, they explained no more than 27.86% of the total variation for any biotic assemblage (Table 3.14). Individual contributions typically were small. As mentioned in the previous section, the significant correlations of "temperature", "salinity", and "dissolved oxygen" with biotic patterns should be viewed with caution given the narrow range of values for these parameters (see Chapter 2 for discussion).

Spatial differences in biotic structure as defined by environmental parameters were apparent for all three epibenthic groups and most evident when viewed at the level of "Area" (Figures 3.9-3.12). Distances among sites/areas were largest for count and percent cover macro-algal groups, with eastern sites clearly different from remaining sites (Figures 3.9, 3.10). Both macro-algal models used a similar number of parameters to explain similar fitted and total percentages of assemblage structure: the count model used 11 parameters to explain 68.86% of fitted and 27.86% of total variation, and the percent cover model used 10 parameters to explain 64.99% of fitted and 22.77% of total variation. The two groups shared two of the top three contributing parameters, but "Habitat" contributed 12% to the counts group and only 7% to the percent cover group. The habitat classes "Boulders, mixed" and "Cobble, mixed" characterised the eastern Flax Bay site for both counts and percent cover. "Harbour proximity" contributed only 6% to both groups. For count and percent cover data, "Harbour proximity" separated central and the western Red Rocks sites.

For percent cover data, the remaining western sites, 2nd Wash and 1st Wash, also separated along this vector, as expected from this distance-based parameter. The third highest contributing parameters were "Hydrology", contributing 7% to the counts group, and "Reef", contributing 12% to percent cover. "Hydrology", defined by the proximity to Karori Rip at the western end of the coast, correlated with assemblage patterns at the three eastern sites. In the percent cover group, "Vertical reef" visibly characterised central and western sites, while "Inclined reef"

characterised the eastern sites. “Inclined reef” correlated with the eastern sites and the western site, 2nd Wash.

The mobile macro-invertebrate assemblages also grouped by area but with the eastern site Barrett Reef lying with the western sites (Figure 3.11). The most parsimonious model used only 7 abiotic variables to explain 58.55% of the fitted variation and 27.23% of the total variation (Table 3.13). In this group, the environmental variables describing physical habitat had the highest correlation with biotic structure, with “Reef” explaining 29% of the variation and “Vertical reef” separating central sites. “Inclined reef” characterised western and eastern sites, while “Horizontal reef” separated these two groups. “Exposure” explained only 4% of the biotic pattern but separated western sites with Barrett Reef from the remaining sites. “Secchi depth” (optical clarity) explained only 2.7% and differentiated eastern sites. Interannual differences were minimal, as evidenced by the nearness of each year’s survey sites.

As with mobile macro-invertebrates, the eastern site Barrett Reef grouped with the western sites for sessile macro-invertebrates (Figure 3.12). This epibenthic group was correlated with twelve environmental variables, none contributing more than 5.0% (“Urban development”) to biotic structure (Table 3.13). This factor separated western sites from most of the central and eastern sites, as did “hydrology” at 2.6% (“Karori Rip yes/no”). “Salinity” explained 2.7% of variation and contributed to the central sites. Environmental variables explained the lowest proportion of total variation (13.62%) for all groups, although the amount of fitted variation explained, (68.5%), was on a par with the other groups.

Table 3.12. Relationships between 13 environmental variables and multivariate patterns in epibenthic assemblages from Wellington New Zealand's south coast. Models were developed using distLM analyses with stepwise selections and Akaike Information Criteria and 9999 permutations.

SEQUENTIAL TESTS for parsimonious models by epibenthic group								
Group/ Environmental variable	AIC	SS (trace)	Pseudo-F	P	Proportion of variance	Cumulative proportion	Residual df	Regression df
Macro-algae counts (subset of 11 species): $R^2 = 0.4046$								
Habitat	1362.1	33739	6.6668	0.0001	0.126	0.126	185	5
Harbour proximity	1349.7	17063	14.468	0.0001	0.064	0.190	184	6
Hydrography	1335.7	17550	16.103	0.0001	0.066	0.255	183	7
Rugosity	1330.1	7830.3	7.4374	0.0001	0.029	0.284	182	8
Reef	1327.3	8669.1	2.8274	0.0003	0.032	0.317	179	11
Urban development	1322.6	6365.8	6.417	0.0001	0.024	0.341	178	12
Depth	1319.9	4312.3	4.4308	0.0013	0.016	0.357	177	13
Temperature	1318.7	2881	2.9935	0.0170	0.011	0.367	176	14
Exposure	1318.2	2156.1	2.2562	0.0579	0.008	0.376	175	15
Secchi depth	1314.5	4995.8	5.3581	0.0002	0.019	0.394	174	16
Dissolved oxygen	1313.2	2799.8	3.038	0.0164	0.010	0.405	173	17
Macro-algae percent cover (all species surveyed): $R^2 = 0.3504$								
Reef	1401	40218	8.589	0.0001	0.122	0.122	186	4
Harbour proximity	1389.6	19821	13.557	0.0001	0.060	0.182	185	5
Habitat	1380.7	22969	4.1991	0.0001	0.069	0.251	181	9
Dissolved oxygen	1373.6	11607	8.8557	0.0001	0.035	0.286	180	10
Depth	1371.5	5091.7	3.9485	0.0014	0.015	0.302	179	11
Salinity	1370.3	3770.1	2.9556	0.0057	0.011	0.313	178	12
Temperature	1369.9	2907.2	2.2957	0.0283	0.009	0.322	177	13
Secchi depth	1369.1	3266.6	2.6029	0.0129	0.010	0.332	176	14
Hydrography	1368.2	3307.1	2.6601	0.0111	0.010	0.342	175	15
Rugosity	1367.7	2842.3	2.3032	0.0272	0.009	0.350	174	16
Mobile macro-invertebrates counts: $R^2 = 0.4651$								
Reef	554.02	41745	9.9962	0.0001	0.29404	0.29404	72	4
Exposure	550.69	6785.6	5.1561	0.0001	4.78E-02	0.34184	71	5
Secchi depth	549.41	3952	3.0913	0.0003	2.78E-02	0.36967	70	6
Urban development	548.58	3262.2	2.6105	0.0032	2.30E-02	0.39265	69	7
Hydrography	548.17	4858.7	2.0004	0.0047	3.42E-02	0.42687	67	9
Temperature	547	3331.9	2.8181	0.0020	2.35E-02	0.45034	66	10
Salinity	546.93	2092.7	1.7912	0.0655	1.47E-02	0.46508	65	11
Sessile macro-invertebrates percent cover: $R^2 = 0.1986$								
Urban development	1445.6	19796	9.9256	0.0001	0.050	0.050	188	2
Salinity	1442.2	10632	5.457	0.0002	0.027	0.077	187	3
Hydrography	1438.7	10319	5.4219	0.0001	0.026	0.103	186	4
Harbour proximity	1436.8	7267.7	3.8777	0.0007	0.018	0.122	185	5
Temperature	1435.8	5333.7	2.8746	0.0054	0.014	0.135	184	6
Dissolved oxygen	1435	4998.1	2.7189	0.0071	0.013	0.148	183	7
Secchi depth	1434.6	4235.3	2.3206	0.0198	0.011	0.159	182	8
Temperature	1434.5	3350.4	1.8357	0.0603	0.008	0.150	183	7
Sediment	1434.2	7523.2	2.0758	0.0088	0.019	0.169	181	9
Rugosity	1433.9	4029.4	2.2388	0.0243	0.010	0.179	180	10
Temperature	1433.6	3902.3	2.1824	0.0259	0.010	0.189	179	11
Exposure	1433.3	3809.7	2.1442	0.0281	0.010	0.199	178	12

Table 3.13. Fitted versus total variation identified by the first two axes of distLM models of relationships between 13 environmental variables and epibenthic assemblages from Wellington, New Zealand's south coast. Models were developed using stepwise selection and Akaike Information Criteria from 9999 permutations.

Group	Fitted variation	Cumulative variation	Total variation	Cumulative variation
Macro-algae counts (subset of 11 species)				
Axis 1	39.26		15.89	
Axis 2	29.60	68.86	11.98	27.86
Macro-algae percent cover (all species surveyed)				
Axis 1	37.22		13.04	
Axis 2	27.22	64.99	9.73	22.77
Mobile macro-invertebrates counts				
Axis 1	40.85		19.00	
Axis 2	17.71	58.55	8.24	27.23
Sessile macro-invertebrates percent cover				
Axis 1	46.68		9.28	
Axis 2	21.82	68.5	4.34	13.62

Figure 3.9. Associations between environmental variables and macro-algal assemblage structure for count and percent cover data **by site**. Vector plot overlays show relationship (vector orientation) and fitted correlation (length of vector) with sites. Models were significant at $p < 0.05$ and developed using distance-based redundancy analyses and the Akaike Information Criteria. Data were collected from Wellington, New Zealand's south coast rocky-reef communities. **Labels are by site to show fine-scale spatial variation.**

Figure 3.10. Associations between environmental variables and macro-algal assemblages (count and percent cover data) presented by **area** as identified by distance-based redundancy analyses. Vector plot overlays show relationships (vector orientation) and fitted correlations (length of vector) contributing to spatial and interannual patterns in biotic assemblages. Models are significant at $p < 0.05$. **Labels are by coastal area to more easily visualize larger-scale spatial patterns.**

Figure 3.11. Associations between environmental variables and mobile macro-invertebrate assemblages identified by distance-based redundancy analyses. Vector plot overlays show relationship (vector orientation) and fitted correlation (length of vector) contributing to spatial and interannual patterns in biotic assemblages. Models are significant at $p < 0.05$. Data were collected from Wellington, New Zealand's south coast rocky-reef assemblages. Both sites and areas are shown to see spatial patterns at two scales.

Figure 3.12. Associations between environmental variables and sessile macro-invertebrate assemblage structure identified by distance-based redundancy analyses. Vector plot overlays show relationship (vector orientation) and fitted correlation (length of vector) contributing to spatial and interannual patterns in biotic assemblages. Models were significant at $p < 0.05$. Data were collected from Wellington, New Zealand's south coast rocky-reef communities. Both sites and areas are shown to see spatial patterns at two scales.

3.3.3 Distribution of species richness and abundance by seabed classes: ANOVAs, Kruskal-Wallis

Tests for significant differences in species richness (total species number) and in density between seabed classes were used to identify the contribution of physical habitat to biotic patterns. Patterns varied for each epibenthic group. Densities and species richness differed significantly among seabed classes for all groups (all $p \leq 0.02$, Table 3.15). Macro-algal counts increased with increasing substratum size and complexity to their highest values for “Broken reef, mixed” (Figure 3.13). Percent cover was also higher for the larger, more complex substrata and the number of species showed a linear increase as well. Mobile macro-invertebrate densities remained low for the smaller mobile classes and also increased for the “Boulder, mixed” class before decreasing for the “Broken reef, mixed” and “Reef” classes. Richness shared a similar pattern for the mobile substrata, but values for this group remained high for the two reef classes. Sessile macro-invertebrate percent cover showed no clear pattern across seabed classes and showed large variation for the smallest class, “Sand/pebble”. However, species richness did show a clear upward trend (non-significant), with the highest value observed for “Reef”.

Figure 3.13. Total number of species and mean density ($\pm SE$) by seabed class for three epibenthic groups from Wellington’s south coast nearshore rocky reefs. Sample sizes are presented below plotted lines and represent total number of sampling units for which data were averaged across both survey years. The seabed complexity classification for “Mobile macro-invertebrates” contains two unique classes: “Algae” and “Overhang”. Data were collected during two annual summer baseline surveys in 2007/08 and 2008/09.

Table 3.14. Differences in species richness and mean densities among seabed classes for three epibenthic groups from Wellington, New Zealand's south coast rocky reefs. One-way analyses of variance and Kruskal-Wallis tests were used to test for significant differences. Bold values indicate significant differences at $p < 0.05$. Data were collected as part of two annual baseline surveys of community structure (2007/08 and 2008/09).

ANOVA tests					
Group	Source	Sum of Squares	df	Mean Square	F p-value
Macro-algae (11 species subset): counts					
<i>Mean number of individuals m^{-2}</i>	Seabed complexity	634.705	4	158.676	2.999 .020
	Error	9840.174	186	52.904	
Macro-algae (full survey): percent cover m^{-2}					
<i>Mean number of species m^{-2}</i>	Seabed complexity	110.658	4	27.664	4.023 .004
	Error	1272.311	185	6.877	
<i>Mean percent cover m^{-2}</i>	Seabed complexity	105848.515	4	26462.129	13.674 < 0.001
	Error	358004.532	185	1935.160	
KRUSKAL-WALLIS tests					
			Chi-square df	p-value	Seabed class Rank
Sessile macro-invertebrates: percent cover					
<i>Mean number of species m^{-2}</i>		14.067	4	0.007	Sand/gravel 68.75 54.25
<i>Mean percent cover m^{-2}</i>		26.151	4	< 0.001	Cobble, mixed 68.88 60.77
					Boulder, mixed 86.52 75.45
					Reef/Boulder/Rubble 82.58 93.23
					Reef 110.22 114.52
Mobile macro-invertebrates: counts					
<i>Mean number of species m^{-2}</i>		112.006	5	< 0.001	Sand/gravel 106.48 103.55
<i>Mean number of individuals m^{-2}</i>		119878	5	< 0.001	Cobble, mixed 157.41 165.82
					Overhang 179.26 176.32
					Boulder, mixed 248.32 267.93
					Reef/Boulder/Rubble 262.19 261.16
					Reef 248.63 241.22

3.4 Aim 4: Comparisons with two previous reserve-related studies

The possible negative interaction between two canopy species suggested by Pande (2000) was tested with a Pearson correlation. No significant correlations were identified between *Ecklonia radiata* and either *Carpophyllum* species using count data per Pande (*C. flexuosum*: $r = -0.013$, $p = 0.855$, $n = 187$; *C. maschalocarpum*: $r = -0.033$, $p = 0.650$, $n = 190$). Her tentative grouping of Barrett Reef with exposed sites was based on its offshore location, but the Baardseth index of exposure used here assigned this site the third-lowest exposure rating of all baseline sites used: a value of 2 versus a mean of 22 for the three western sites (Table 3.1).

The high degree of patchiness identified by Cole and Russell (1989) for paua (both *Haliotis* species grouped together) and kina was revisited. They obtained, based on 1m² quadrats rather than the belt transects used here, a variance/mean ratio of 3.35 and percentage of zero counts = 75% for paua. For kina, these values were 25.7 and 85.5%, respectively. Here, I found for *Haliotis australis* a variance/mean = 0.12 with zero-counts = 32.8%; for *H. iris* a variance/mean = 0.14 and zero-counts = 44.7%; and for kina a variance/mean = 0.19 and zero-counts = 13.1% when densities were averaged to m² for comparison with the earlier study. Although Cole and Russell (1989) did not present values for rock lobster, the variance/mean value here was 0.11 and zero-counts were 55.3%.

4. Discussion

Coastal resource managers must formulate management strategies to address a range of negative impacts on ecosystem structure, diversity, and function that include habitat fragmentation, degradation, and destruction; pollution, eutrophication, increased and stronger storms, ocean warming, rising sea levels, and competing stakeholder interests. The development of such resource management policies requires an understanding of pre-protection natural patterns in the biotic and abiotic components of communities that is typically provided by baseline assessments of the species, community, or area(s) of interest. Knowledge of the pre-impact community structure and status can then be used to develop time- and cost-effective

long-term policy and management protocols that can preserve communities at the desired level of structure and function.

Choosing field protocols can be challenging when developing baseline and long-term monitoring plans. The baseline and monitoring studies of soft- and hard-bottom communities publically available from New Zealand's Department of Conservation (www.doc.govt.nz) describe a range of sampling approaches that include different survey areas, sampling units, target species, methods of site selection, and techniques (e.g. fixed or randomly distributed quadrats; permanent and random sites; visual, photo surveys, remote or swum video recording).

This baseline study of WSC nearshore rocky-reef epibenthic assemblages has provided the first broad-scale baseline assessment of spatial and interannual variation in abundance, size-class, diversity (Chapter 4), and size frequency patterns (Chapter 5) of a broad group of commonly-occurring epibenthic species. It is the first WSC study to quantify substratum heterogeneity and describe patterns in substratum composition, profile, and distribution and incorporate these into multivariate models of community structure. It is also the first WSC study to identify, through community composition analyses, representative subsets of species and to provide updated field survey methods for long-term monitoring of community status in response to natural and anthropogenic (marine reserve, climate change) effects. Finally, these surveys' direct observations have served as ground-truthing for the remote mapping component of this project (chapter 7).

Specifically, this study identified significant spatial and interannual patterns in community structure for three epibenthic groups based on abundance metrics and identified environmental correlates contributing to assemblage patterns. Although spatial components to composition and distribution were identified, they did not follow a previously identified linear east-west gradient. Moreover, these patterns varied among epibenthic groups. Abundance and size-class patterns varied among species within each group. Numbers of many mobile macro-invertebrates surveyed increased at the central sites. *Ecklonia radiata* was the only canopy species present at all sites in both years. The invasive species *Undaria pinnatifida*, present in high numbers in Wellington harbour and along exposed southern New Zealand coasts (Russell et al. 2008), was present only at very low densities at shallower depths at only

two sites, suggesting that abundances were not yet high enough to threaten native canopy. Identifying structural redundancies in each epibenthic group reduced the number of species needed in future monitoring of natural and anthropogenic changes to a more manageable number.

4.1 Epibenthic community patterns

Temperate hard-bottom kelp forest seascapes, dominated by members of the orders Fucales and Laminariales, are characterised by spatial patchiness and mosaic composition at local to regional scales (Steneck et al. 2002, Miller et al. 2009). Kelp forests are essential to temperate coastal systems because of the range of services they provide and their role in physically structuring the environment (Dayton 1985a). Monitoring kelp forest status is essential because of their susceptibility to eutrophication, altered salinity levels, increased storm frequency and intensity, increased sea temperatures, ocean acidification, and habitat modification (Steneck et al. 2002). Mosaic composition and density reflect complex direct and indirect interactions between three categories of biological and physical processes: recruitment, growth, and competition (Lubchenco and Gaines 1981, Steneck et al. 2002). Biological contributions to rocky-reef communities include primary and secondary productivity, refugia for invertebrate and vertebrate fauna, and contributions to nutrient cycles from detritus and drift algae. Kelps physically structure biotic and abiotic habitat components through shading, sweeping, and bioerosion of substratum (Smith and Bayliss-Smith 1998, Garden and Smith 2011). By modifying wave action and current flow, they alter rates at which nutrients, sediment, and larvae depose (Eckman et al. 1989). Existing patterns of abundance, diversity, and distribution are due partly to prior interactions between larval supply, transport and recruitment with periodic and stochastic hydrodynamic and meteorological processes (Eckman et al. 1989).

The nature of trophic interactions on kelp forest composition varies regionally. Top-down control of macro-algal composition characterises certain temperate regions, with apex predators such as otters structuring kelp forests indirectly through consumption of primary consumers (fish), allowing increases in urchin populations

that create algal “barrens”, or directly through fish grazing on macro-algae (Dayton et al. 1984, Dayton 1985b). Within New Zealand, a study found that this top-down control by predators that is a primary structuring process in other temperate regions is not the dominant process structuring kelp forests here. Two exceptions include the well-documented urchin barrens related to reserve status here in northern New Zealand’s northern Leigh Marine Reserve (Ayling, 1981) Shears & Babcock, 2003; Parsons, 2004). In southern New Zealand, the conditions in which the fish foraging creating urchin barrens there were minimal resemble WSC conditions: high-density, high-exposure areas (Schiel 1988; Villouta et al., 2001). The similarity of coastal conditions, coupled with WSC’s historically smaller urchin populations, suggest that kina lack the population to drive this density-dependent process. Rather, composition here appears to be driven by resource competition and interactions among macro-algae (Schiel and Hickford 2001).

4.2 Macro-invertebrate patterns

Macro-invertebrates other than species harvested prior to reserve status should be included in baseline and monitoring programs because of their contribution to all functional groups: herbivore, carnivore, detritivore, omnivore, and filter-feeder. As cosmopolitan contributors to ecosystem functioning, changes in their status can signal impending or actual changes to the environment. Not only kina but other mobile macro-invertebrates have been shown to structure macro-algal communities through grazing. The gastropods *Turbo smaragdus* and *Cookia sulcata* were found to clear subtidal macro-algal patches locally (Ayling 1981, Creese 1988), while *Turbo* spp. in East Australia removed canopy (Baker et al. 2008). Sessile macro-invertebrates, including Ascidians, Bryzoans, Hydrozoans, Poriferans, and Serpulids, structure physical habitat and assemblages through the formation of biogenic reefs (lacking on WSC), benthic-pelagic coupling, and provision of microrefugia, and competition for space, a limiting resource in patchily distributed rocky-reef habitats. These groups also contribute to patch composition by modifying wave impacts and nutrient and larval deposition through the alteration of boundary-layer flow (Bell 2008, Davis 2009). Although sessile invertebrates typically populate vertical reef surfaces, on WSC they were only observed in small patches on both horizontal and vertical surfaces. No

differences to this distribution or limited cover were observed on recreational dives to 30+ metres.

Long-term patterns of low abundance of paua and kina at the western sites recorded here and by (Pande & Gardner, 2009), coupled with skewed size-frequency distributions, suggest that these western populations have limited recruitment and are too small and individuals too dispersed to self-sustain, as predicted by the Allee effect (Stephens and Sutherland 1999). Cole and Jackson's 1989 suggestion that paua numbers might rise in the future has not been born out. Low abundances at these sites might be due to the limited distances travelled by paua larvae, limiting the ability of local populations to self-recruit (McShane et al. 1994). This limited dispersal characterises kina larval stages as well, with dispersion, recruitment, and post-settlement survival rates affected by hydrodynamic processes and storms that can lead to offshore advection (Lamare 1998). Predation, grazing, and space are other factors affecting macro-invertebrate distribution. Additional environmental factors affecting juvenile paua survival included shallow depths and greater wave exposure, with proximity of conspecific adults contributing to juvenile mortality (McShane and Naylor 1995, Naylor and McShane 2001). The very low numbers of rock lobsters observed here did not match reported landings for commercial harvests for this area: the corresponding catch for commercial area #915 was ~79 tonnes (Sykes 2012). Clearly the sampling method employed here was inadequate to the depth, habitat, and diel patterns of this species (see below for recommendations).

Crustose (encrusting) coralline algae should be included in long-term monitoring because of its potential role as an indicator group of environmental stress and its role in invertebrate recruitment. As a carbonate-fixing organism, its growth was experimentally shown to be negatively affected by the higher levels of dissolved carbon dioxide associated with ocean acidification (Kuffner et al. 2008). The group was found to play a role in larval settlement in two ways: chemically, through cues released that trigger paua and kina larval settlement (Lamare and Barker 2001, Roberts et al. 2004, Naylor 2006), and morphologically, with rugose taxa attracting greater numbers of larvae in a South African abalone recruitment experiment (Day and Branch 2002).

The continuity of abundance, distribution, composition and congruence of macro-algal and macro-invertebrate size-classes with previous studies (Cole and Jackson 1989, Schiel 1990, Pande 2001, Shears and Babcock 2007) indicates some degree of long-term stability of WSC nearshore rocky-reef communities. Long-term stability does not preclude natural variability in abundance or distribution, noted here in slight differences in canopy and macro-invertebrate (kina, paua) abundance and distribution patterns between the previous baseline survey and this. Such temporal and spatial variation in abundances is a fundamental characteristic of rocky-reef kelp-forest communities (Dayton et al. 1984, Dayton et al. 1992, Miller et al. 2009). SIMPER identified canopy species as the primary drivers of intra- and inter-site differences but did not identify as important the presence in larger numbers of fleshy understorey red algae in Breaker Bay despite this group's obvious presence at this location.

4.3 Size-class distribution patterns

Size-frequency distributions provide useful information for managers on demography, spawning activity, recruitment, growth, mortality, and harvesting (Valiela 1995, Pomeroy et al. 2004). It complements abundance estimates to provide information on population structure and biological processes, and possible environmental stress. Size-class can serve as a proxy for age-structure because methods for estimating age of benthic organisms are lacking for some groups (Kingsford and Battershill 1998). The shape of frequency distributions is affected by disease, predation, mortality, and harvesting. Shifts in number of classes or missing classes can indicate changes in one (or more) physical variables, or anthropogenic pressure such as harvesting (Carignan and Villard 2002). Skewed distributions often reflect small populations (Ayling 1981). Small populations can be more vulnerable to natural and human disturbances and processes, thus less able to recover and persist. Evenly distributed size classes for species in a protected area can indicate successful management: such class distribution can indicate that spawning, recruitment, growth, and maturity processes are intact. Collection of size data can form part of an ecosystem status assessment when additional information on health such as colour, damage, and morphology are included (Pomeroy et al. 2004). The size classes of studied species observed here are characterized by skewed distributions and missing

classes, both of which correspond to distributions noted in previous studies (Cole and Jackson 1989, Pande 2001, Shears and Babcock 2007). It is likely that these distributions reflect local hydrodynamic processes and meteorological events, their interaction with spawning activity, and predation.

4.4 The use of parataxonomy and indicator species

Consistent, reliable and discriminating identification of members of such morphologically disparate groups as various sessile invertebrates and morphologically diverse macro-algae can be challenging to long-term replication of baseline and monitoring surveys. Species-level identification is the most commonly used for biodiversity and monitoring studies and provides a universal basis for inter-study comparisons, affording the option to scale up taxonomically (Carignan and Villard 2002). Still, surveying at this level of taxonomic resolution is not always possible when limited or no descriptions of organisms are available or when, as is often the case, non-expert surveyors are used. Other factors to consider when planning dived surveys include time and cost, when survey sites are remote and diveable conditions often limited. Options for rapid methods for describing biodiversity and quantifying abundance include keying organisms to higher taxonomic levels; this approach was successfully used in the development of the New Zealand Ministry of the Environment's marine environmental classification (MEC) scheme (Snelder et al. 2005). Conversely, inter-regional differences in intertidal algal communities off the Pacific Northwest were not well resolved when using order and above levels of taxonomy (Konar and Iken 2009).

A second option is the use of parataxonomy, the classification of organisms into observational taxonomic units based on morphotype (Obrist and Duelli 2010). This approach was an effective, informative and replicable method when used for assessing resources in a range of habitats: tropical sponge communities (Bell and Barnes 2001, Bell et al. 2006), invertebrates in three terrestrial environments, and intertidal macro-algal assemblages (Konar and Iken 2009). Despite these successes, consistent and replicable classification of flora based on morphotypes was found to vary among volunteers, and detection of habitat use was not as powerful as when assessed using Linnaean identification (Abadie et al. 2008). Although clearly defined definitions of

morphotypes ensure consistency and continuity, one drawback to this approach can be the specificity of morphotype definitions to location, project, and observers. Also, spatial scale was found to affect relevance, with morphometric features identifiable or present at small scales not visible or present at larger scales (Nipperess et al. 2008).

A third alternative to the problem of field identification is the use of focal species, comprising indicator, flagship, keystone, and umbrella species. Here, a subset of species to be used for assessing conservation goals and ecosystem functioning is selected for their ability to model higher-order assemblage complexity and status, or to signal impending or actual biological or environmental change (Zacharias and Roff 2001). Criteria to consider when selecting species for their ability to model status of and changes to higher-order patterns, rather than their propaganda, cultural, or economic value, include: 1) rarity; rare species do not make good indicators (Fleishman et al. 2000); 2) ease and accuracy of identification; 3) temporal and spatial persistence; and 4) response to negative impacts at an early, remediable stage (Dale and Beyeler 2001). This approach has its pros and cons, as seen in the transition from the historical single-species resource management paradigm to the ecosystem-based approach (Simberloff 1997). One problematic assumption of the focal-species concept is that selected species adequately represent the needs (e.g. light, nutrients, space, substratum) of those they model. The Gaussian niche theory that two species cannot share the same resource for extended periods suggests that factors negatively affecting resources used by an indicator species might not necessarily affect those used by a modelled species. Thus the use of focal species assumes a mean response to stress when the ability to adjust can be variable across groups (Wahl 2009). A second criticism of this approach is that the selection of focal species can be biased (Carignan and Villard 2002).

This study analysed community structure at the species level for continuity with past and future studies and to permit comparison with similar local studies. Classification by morphotypes of sessile macro-invertebrates was consistent among sites and between years, but omitted rarer and less abundant species (see suggestion, below). The nature of the species subsets recommended here for future monitoring does not correspond to the traditional definitions of the four types of focal species; they simply reflect intra- and inter-site composition so “indicate” compositional

patterns. They are suitable because they meet the criteria of spatial and temporal persistence, they can be easily and consistently identified by non-experts, and the use of a statistical routine to select representative species avoids bias. A surprising result was the number of non-traditional mobile macro-invertebrate species identified as representative by BEST analysis. What is not known about the representative species selected here are their threshold responses to biological and environmental stressors. This is a topic requiring further study.

4.5 Environmental contributions to assemblage patterns

The overall low correlation values were surprising, as were the significant contributions of three water parameters with very small ranges in values. The lack of congruence between the MDS plots for environmental and biotic models and the relatively low correlation of the environmental variables included here suggest that either WSC communities are structured more by biological interactions, the environmental variables used here were not sampled at the range at which they could impact the assemblage, the sampling scale was too small, or more informative predictive variables were not included. The generally low correlation values were unexpected. Although the correlations of “salinity”, “temperature”, and “dissolved oxygen” were low, their significance was bewildering in view of the small range in values and similarity across sites. For example, elsewhere a freshwater lens did not deter the growth of northern temperate canopy species (Dayton 1985a). Perhaps these three variables are a proxy for an unidentified variable. The extremely low (but significant) contribution of “exposure” and “rugosity” is hard to explain, although “rugosity” is one aspect of “substratum”, levels of which were more highly correlated with certain groups. It is possible that the method of calculating exposure used here did not capture WSC patterns.

The clear east-west gradient in three water parameters along WSC, with lower values at the western end (Oteranga Bay) (Helson et al. 2007) was not identified by this study's suite of environmental factors. Although this lack of concordance might be due to the different water parameters measured in Helson's study (nutrients, particulates, and productivity), it was expected that the underlying hydrological

processes determining their distribution would have led to a similar east-west gradient in this study's environmental data.

The inclusion of three environmental variables ("Urban development", "Hydrology", and "Distance from harbour") defined by an east-west gradient to identify east-west gradients could be an environmental tautology. Certainly these three spatially-defined variables dominated the correlations and MDS ordinations and different patterns emerged after their exclusion. These three variables were intended to model aspects of coastal geography, hydrodynamic patterns, and urban development. It was expected that at least one of these would capture the effect of possible poaching at the western end on paua populations but this was not the case. Experimentation with analyses using each of these three variables separately did not change the outcome of the analyses.

Future studies might consider these variables at different scales, different variables entirely, or the nature of biological interactions in determining WSC epibenthic composition. Certainly environmental variables contribute to community mosaics: hydrodynamics, wave stress, surge, light, temperature, sediment, competition for space and substratum type and orientation are among the physical variables affecting reproductive success, recruitment, growth, mortality, and persistence (Connell 2003, Grantham et al. 2003, Booth and Ayers 2005, Connell 2005, Phillips and Shima 2006, Lucieer and Pederson 2008, Toohey and Kendrick 2008, Scheibling et al. 2009).

4.6 Recommendations

- Annual summer monitoring at nine sites, excluding Flax Bay. While seasonal surveys would provide a more refined understanding of WSC benthic phenology, despite a previous study's recommendation for these, completing distinct seasonal surveys consistently was found to be extremely difficult. The difficulty of sourcing and training divers, volunteer or paid, and assembling full dive teams when sea and weather conditions unexpectedly permit fieldwork is a resource-intensive, frustrating task. The congruence of patterns identified by previous WSC surveys and this work suggests that

annual surveys can provide a working, if less comprehensive, understanding of ecosystem status.

- Use species subsets for epibenthic groups; include *Caulerpa* and CCA.
- Retain changes to quadrat and belt transect size.
- Continue use of randomly distributed quadrats and transects.
- Conduct future surveys using GPS coordinates as hubs in combination with random headings to minimise background variation.
- Deeper community surveys: Because the depth of the marine reserve extends to ~40m, monitoring surveys should be extended to deeper communities. Non-survey dives along the extent of WSC noted that deeper communities were dominated by Rhodophyta and sessile-invertebrates.
- Retain all nine sites to maximise statistical robustness and control for natural spatial variation. All three western sites lie in “the wind factory” area; once Red Rocks is reached then the remaining two sites are quickly reached.
- Dedicated and separate monitoring teams and/or methods for:
 - Rock lobster: Dedicated sampling at greater depths over soft and hard-bottom areas using CPUE pot data or, if in shallow areas, larger belt transects (Kelly et al. 1999, Booth 2003, Stewart and MacDiarmid 2003, Shears et al. 2006).
 - Paua: nocturnal observations could be incorporated to identify smaller size-class abundances. Given the logistical challenges of night diving along WSC, dawn or dusk collections could be an acceptable alternative choice.
 - Sessile macro-invertebrates: Dedicated sampling using a gridded 0.1m²-0.25m² frame and image analysis software, which will facilitate far more accurate biodiversity and density estimates and finer taxonomic resolution than can be obtained visually in WSC's typically surge-y subtidal zone (Bullimore 2001).
 - *Macrocystis pyrifera*: Based on observational data, this species was underestimated in these surveys because its preferred habitat appears to

be smaller mobile substrata, which fell outside the area of hard-bottom substrata targeted in these surveys. Because this is a keystone species in northern temperate coastal systems (Dayton 1985a, Steneck et al. 2002) with limited distribution in New Zealand (Schiel 1990), this would provide new information on community dynamics locally.

- Water quality monitoring: A regular sampling program should include variables such as: salinity, temperature, dissolved oxygen, water clarity (Secchi depth (quick and inexpensive) or particulate matter (time and cost involved), chlorophyll *a*, dissolved organic matter, total particulate matter, % carbon, % nitrogen, and heavy metals. Such work would build on previous a WSC water chemistry study (Helson et al., 2007), which would provide information on long-term patterns that could help management in monitoring and planning coastal development in the area. Only three sites along the length of WSC would provide adequate coverage of natural processes and human activity in highly developed (eastern), transition (central) and rural (western) areas, reflecting the range of human presence along WSC.

Chapter 4. Diversity of Wellington's epibenthic rocky-reef assemblages

1. Introduction

Biodiversity provides quantitative, qualitative, and taxonomic descriptions of the biological communities that drive the planet's biogeochemical cycling (Hooper et al., 2005; Kunin & Lawton, 1996; Magurran, 2004). It is defined at three levels of biological and ecological structure: genetic, species, and ecosystem (Gray, 1997). The benefits of diversity at each level include on-going fitness, adaptability, resilience to disease and invasions, and provision of ecosystem goods and services (Hooper et al., 2005). Genetic diversity confers advantages to future generations through adaptation and evolution, while species diversity can protect against total loss of productivity through functional redundancy, the ability of one or more species to perform another species' functions regardless of taxonomic relatedness. Diverse communities therefore have greater potential resilience to withstand negative impacts. Ecosystem diversity is the composite of biological and physical diversity within an area defined by scientific, biological, managerial, economic, cultural, or political interests. The concept recognises the role of physical habitat type and distribution and environmental processes in structuring communities, in addition to biological processes (e.g. settlement, recruitment, resource competition, predation) (Fraschetti et al., 2008; Thrush et al., 2011). Ecosystems recognised as unique for their diversity in addition to their functioning include temperate kelp forests, tropical coral reefs, oceanic plankton communities and upwelling coastal ecosystems. (Costello et al., 2010; Ray, 1991; Ray & McCormick-Ray, 2004).

Biodiversity is defined at five spatial scales: point, alpha (local), beta (area), gamma (region), and omega (phylogenetic). Theories explaining biodiversity patterns reference biological and physical processes at a range of spatial and temporal scales that are hierarchical: higher-level phylogenetic factors structure smaller-scale ecological interactions and assemblage structure (Ray, 1991). Spatial diversity patterns include latitudinal, longitudinal, and elevation gradients, species-area relationships; disturbance responses; character-displacement effects and primary productivity-linked effects. Temporal scales at which diversity shifts occur are evolutionary or ecological,

with successional and seasonal mechanisms defining the latter (Connell, 1979; Huston, 1979; Wilson, 1994; Rosenzweig, 1995). At the local scale, diversity patterns are a function of past events and current density-dependent and independent processes. Historical and on-going disturbances, resource availability, predation, disease, consumer abundance, substratum quality, and environmental quality interact with r- and k-strategists to determine current levels of species richness and abundance (Huston, 1979; Valiela, 1995). Covich's (2004) review of experimental local-scale studies on the links between diversity shifts and local benthic processes found that the nature of such changes varied and were species-specific. However, low-diversity ecosystems do not necessarily reflect low functional status; areas such as the Arctic Sea and upwelling plankton ecosystems are characterised by low diversity but are considered sustainable with high levels of productivity (Suchanek, 1994).

Biodiversity in the ocean is phylogenetically richer than terrestrial systems, with the ocean containing 34 of the 35 existing phyla. Although the ecological and evolutionary processes that structure diversity in terrestrial and marine environments share similarities, the scales at which the processes operate differ (Carr et al., 2003). The coastal zone is also impacted by unique physical and geochemical processes (Ray, 1991). Physical processes that are unique to the ocean environment are water's properties, which create 3-dimensional environments with unique light, temperature, saline, density, oxygen, drag and flow gradients. Another unique feature of marine assemblages are the open populations; these tend towards higher genetic diversity and less spatial structure. Dispersal ranges of benthic functional groups appear to be group-specific and larger than terrestrial counterparts. Other ecological differences between marine and terrestrial systems with the potential to influence diversity patterns include rates of anthropogenic material transport, life-history rates, population structure responsiveness, and settlement and recruitment processes (Carr et al., 2003; Kinlan & Gaines, 2003)

The use of biodiversity values in developing conservation protocols for impacted species and habitats is often inseparable from aesthetic, historical, cultural, and ethical interpretations of these values (Kunin & Lawton, 1996). Separating their functional value from stakeholders' emotional interpretations is not always possible when conservation priorities are driven by private-interest organisations in this era of limited funding. That

conservation agendas can be driven by public perception of a species or habitat's functional value and irreplaceability is a concern for conservation agencies and governments when using biodiversity indices as one of the tools for setting management priorities (Miller, 2008).

Biodiversity indices, despite their shortcomings (Hurlbert, 1971), can be useful tools for resource managers when used with an understanding of these limitations, because they provide clear measures of community status, uniqueness, and biological relatedness of patchily distributed habitat. Understanding the relationship between diversity and the underlying bio-physical drivers helps managers to identify and prioritise which biotic and abiotic resources require protection, at what scale protection measures should be implemented (Ray & McCormick-Ray, 2004). Genetic and species diversity can be supported by a well-designed marine reserve network that incorporates existing knowledge of local and regional-scale transport corridors (Bell & Okamura, 2005) Ray, 1991; (Carr et al., 2003; Ray & McCormick-Ray, 2004). Diversity values can also, in combination with other biological and geo-physical (and social) indices, be used as input to develop biological valuation maps for policy development and monitoring (Derous et al., 2007). Estimates of the effects of reduced habitat diversity and biodiversity on ecosystem services degraded by anthropogenic activity can be developed through comparisons with studies on functional redundancy in similar ecosystems. Such comparisons, however, are constrained by the specificity of functional values to complex interactions specific to a location (Hooper et al., 2005).

Biodiversity is assessed by partitioning abundance into species richness, equitability, dominance, rarity, and taxonomic relatedness, with many indices developed to accommodate data type, researcher assumptions, and collection method (Humphries et al., 1995). β -diversity indices identify directional change or turnover, spatial change in assemblage composition; and non-directional change, the taxonomic heterogeneity within assemblages (Anderson, M.J. et al., 2011; Anderson, M. J. et al., 2006) Magurran, 2004; R M Seaby & Henderson, 2006). Partitioning variability in assemblage structure at multiple spatial scales allows the development of hypotheses about the underlying environmental mechanisms for these patterns (Crist et al., 2003; Hewitt et al., 2005). However, no derived index is without bias, with each weighting abundance and richness differently. Therefore, comparisons among indices and studies

can be problematic and must be made with an awareness of their inherent biases (Hurlbert, 1971; M J Anderson et al., 2011).

Indices have been used in a range of marine resource management and conservation projects: development of a New Zealand marine classification system (Gordon et al., 2010); descriptions of difference in fish assemblages among temperate New Zealand fish assemblages (Williams et al., 2008); assessment of post-disturbance recruitment processes in temperate New Zealand sessile macro-invertebrate assemblages (Smith & Witman, 1999); development of abiotic surrogates to identify endangered habitats and populations and processes structuring diversity (Thrush et al., 2006; Arias-Gonzalez et al., 2008; McArthur et al., 2010); shifts in functional group dominance due to eutrophication in a European coastal system (Worm et al., 1999), and identification of pollution on benthic community structure in the North Sea (Warwick & Clarke, 1994, 1995).

1.1 Chapter Aims

Identifying patterns and boundaries in the taxonomic composition of local communities contributes to an understanding of the scale at which species meet their resource requirements. This knowledge helps conservation agencies to identify and protect critical habitat and associated environmental processes at appropriate spatial scales. Information on local-scale biodiversity and community turnover serves another management purpose related to the recently created Taputeranga Marine Reserve: local-scale biodiversity values can serve as a benchmark against which reserve-related changes in assemblage structure can be evaluated. Knowledge of how reserve and adjacent areas respond to protection will contribute to the current development of New Zealand's national network of marine reserves.

In this study my aims were to provide resource managers with a benchmark description of local biodiversity patterns, taxonomic characteristics, spatial patterns for two and relationships between two groups of Wellington south coast's rocky-reef epibenthic assemblages: macro-algae (MA) and mobile macro-invertebrates (MMI). The sessile macro-invertebrate group included in the two annual pre-reserve baseline surveys (Chapter 3) were not included here because the lack of available taxonomic expertise limited the development of a full species list. Specifically, I examined site

diversity in terms of richness, evenness, and dominance with univariate indices; correlations in diversity between the two functional groups; sampling adequacy with species-accumulation curves; and spatial patterns (turnover) in homogeneity and taxonomic characteristics using β -diversity analyses. This information complemented the assemblage descriptions based on abundance and size-frequency (chapter 3) and biomass (chapter 5).

2. Methods

This study used data from ten sites surveyed for two annual baseline surveys of Wellington south coast's nearshore rocky-reef kelp assemblages (see chapters 1 and 3 for the area description, survey methodology and list of taxa surveyed). The two benthic groups for which full taxonomic identifications were made were assessed here: macro-algae (MA) (percent cover, 30 species) and (MMI) mobile macro-invertebrates (counts, 36 species). Eight macro-algal species identified to morphotype in chapter 3 were not included to allow the use of taxonomic diversity analyses.

To identify possible spatial trends, comparisons among sites were made on data averaged by site and for both survey years (2008-09). Statistical analyses were conducted using PRIMER, Species Diversity and Richness-IV, and SPSS. The univariate diversity indices calculated here to identify patterns in raw and derived descriptions of richness and evenness were: total number of species (S); total number of individuals/percent cover (N); the complement of Simpson's index ($1-\lambda'$) to describe species richness; Pielou's index (J) to describe evenness; and the Berger-Parker index reciprocal ($1/d$) to describe dominance. The Berger-Parker index highlights the contribution of the most abundant species to diversity while factoring in evenness. The Simpson's index ($1-\gamma'$) is a preferred index because of its independence from sampling effort (Magurran, 2004). Spearman's rank-order correlations were calculated to assess the relationships for species richness and community evenness between MA and MMI. Data were assessed for test assumptions (normality, linearity, homoscedasticity) before conducting one-way between subjects analyses of variance (ANOVA) on each index to compare values among site with alpha set = 0.05. To identify the location of significant differences among sites, the post-hoc Tukey's HSD routine was selected to minimise inflated Type I error rates (Kachigan, 1986).

Visual assessments were made with rank-abundance plots to summarise distributions of species richness and evenness. Coleman's routine verified the assumption of sample homogeneity for estimating total species richness by plotting species accumulation curves from randomised observed samples of each group (R.M. Seaby & Henderson, 2007). Total species richness for each group was then estimated using several methods to ensure accurate total species richness estimates (Magurran, 2004). The methods selected were chosen because they offered the least unbiased estimations (Chao, Jackknife, and Bootstrap routines, 999 permutations) (R.M. Seaby & Henderson, 2007). The adequacy of sampling effort for each epibenthic group was assessed by the presence of asymptotic limits in each group's species accumulation curves.

β -diversity along the south coast was assessed using the PERMDISP routine. This routine was selected because it included post-hoc pairwise comparisons to identify spatial trends. PERMDISP was preferred over Whittaker's index because the site-specific values generated permit the identification of spatial trends. Non-metric multi-dimensional scaling (MDS) plots presented spatial patterns in diversity in a visual format. Vector overlay plots presented Spearman rank correlations of species with multivariate assemblage patterns (M. J. Anderson et al., 2006). Species with correlations higher than 0.5% and 0.7% were marked with a "*" or "_**_" respectively. These values were arbitrarily defined as boundaries based on the results from both groups.

The average taxonomic distinctness ($\Delta+$), AvTD, was used to identify the relatedness of baseline site assemblages. The advantage of this routine is its suitability for studies conducted using different sampling protocols, which is a feature allowing the comparison of baseline and monitoring results from other regions or different agencies that have used different survey protocols. The routine is also preferred because its orthogonality to species richness and evenness provides an independent third component to deconstructing diversity patterns. Samples were viewed in funnel plots with 95% limits representing the expected taxonomic range. This expected range was defined by randomly sampling a master taxonomic list compiled from all taxa observed at all sites during both annual surveys (Clarke & Warwick, 2001).

3. Results

Significant differences between sites were identified in both groups for several of the indices (Table 4.1, Figures 4.1, 4.2). Differences in values between sites were greater for raw indices than for the derived values. Spatial trends in raw and derived univariate indices differed between the macro-algal (MA) and mobile macro-invertebrate (MMI) groups.

3.1 Alpha diversity: raw indices

For the MA group, S differed significantly among sites ($p < 0.001$) but the range was narrow (4-7.7 species) (Table 4.1, Figures 4.1-4.2). The highest value occurred centrally at Yung Pen (7.7 species) and the lowest values at the three eastern sites (4.0-4.5 species). A different spatial pattern characterised abundances, which also differed significantly ($p = 0.004$). Here, the highest values were observed at the two westernmost sites (84.2%, 90.5%), with lower, similar values at the remaining sites. The lowest percent cover occurred at the central site, The Sirens (54.3%). Again, spatial patterns differed for the MMI group: the lowest values for S (7.1, 7.5) and abundance (24.3, 34.0) both occurred at the two westernmost sites ($p < 0.001$). The numbers of species were highest towards the eastern end of the coast (12.3-12.8) before decreasing slightly at the harbour entrance site (10.5). Unlike S , abundance for MMI remained low at most sites, with higher values observed at four eastern sites (maximum of 205.1 at Flax Bay) before falling to 47.0 at the harbour site.

Macro-algal species richness (Simpson's index) was not correlated with mobile macro-invertebrate richness ($\rho = 0.140$, $n = 19$, $p = 0.567$). Macro-algal community evenness (Pielou's J') was also not correlated with mobile macro-invertebrate evenness ($\rho = 0.510$, $n = 19$, $p = 0.836$) (Figure 4.3). These results indicated that changes in richness and evenness values in one group were not related to changes in the other group.

3.2 Alpha diversity: derived indices

For MA, Simpson's index values differed significantly between sites ($p = 0.002$) and identified similar patterns in species richness, with lower values observed at the eastern end of the coast (harbour entrance) (Figures 4.1, 4.2). Although evenness did not differ significantly ($p = 0.680$), Pielou's J was also lower at this end of the shore. No

differences in patterns of dominance, described by the Berger-Parker index, were identified for this group or MMI ($p \geq 0.099$). MMI richness as described by Simpson's index was similar across the coast ($p = 0.060$) but evenness did differ ($p = 0.009$). This variation in Pielou's J values did not follow a linear spatial gradient, with lower values observed at western and central sites and higher values observed at western, central, and the harbour (easternmost) site. Again, it should be noted that the sizes of the differences in the derived indices were not large (~ 0.2).

Table 4.1. One-way analyses of variance testing for differences among survey sites for each diversity index. Both functional groups differed significantly in richness and evenness as measured by multiple metrics. Data were collected off Wellington, New Zealand's south coast (summers, 2007-8 and 2008-09). **Bolded** values indicate significance at $p < 0.05$.

Epibenthic group	Source of variation (Diversity index)	df	MS	F	p
Macro-algae	Species richness	9	26.050	7.283	<0.001
	Number of individuals/Percent cover	9	3013.712	2.814	0.004
	Berger-Parker $1/D$	9	0.950	1.810	0.069
	Simpson's $1-\lambda'$	9	0.581	3.079	0.002
	Pielou's J	9	0.850	1.819	0.680
	Shannon $H \log_e$	9	0.097	2.042	0.037
Mobile macro-invertebrates	Species richness	9	30.547	5.094	<0.001
	Number of individuals/Percent cover	9	26279.920	18.301	<0.001
	Berger-Parker $1/D$	9	1.260	1.731	0.099
	Simpson's $1-\lambda'$	9	0.032	1.947	0.060
	Pielou's J	9	0.041	2.735	0.009
	Shannon $H \log_e$	9	0.250	2.540	0.014

3.3 Visual assessments of sampling adequacy

Sample heterogeneity was minimal for both MA and MMI as identified by the Coleman routine (Figure 4.4, top 2 rows). Heterogeneity was identified by differences in slope between the randomised and theoretical sampling curves, but here the differences were minimal, with close agreement of the shapes, scales, and slopes of each group's curves. The concordance of the estimated total species richness curves, $S_{\max}$ (Figure 4.4, bottom row) with S_{observed} indicated that most of the epibenthic rocky-reef diversity of each group was captured by the end of the first year's baseline survey when asymptotic limits for both groups were approached at the corresponding

number of samples (MA = 90; MMI = 40). Although total species richness as estimated by jackknife and bootstrap techniques was greater than observed values for MMI, the difference between the curves was not large. This agreement between S_{observed} and estimated S_{max} values suggests that these baseline surveys adequately captured the biological characteristics of shallow kelp forest assemblages along Wellington's south coast and that additional sampling would not contribute more information.

Rank abundance curves (Figure 4.5A) described similar patterns in species richness and evenness for both groups. Abundances for both were dominated by a small group of common species (Y-axes relative abundance intercepts) and these abundances were unevenly distributed among all species (steep slopes). However, diversity was greater among MMI, with eastern sites (green lines) having slightly higher richness (larger X-axes intercepts) and evenness (shallower slopes) values relative to the other sites/areas and to MA. These patterns corresponded to the plotted Simpson's and Shannon's values discussed above.

3.4 Taxonomic distinctness

Taxonomic distinctness ($\Delta+$) patterns differed between MA and MMI (Figure 4.5 B). In both epibenthic groups spatial trends were present but these did not conform to a linear-west east gradient. MA assemblage composition was most heterogeneous for the eastern sites, with $\Delta+$ ranging from 66.9 at Breaker Bay to 84.4 at Palmer Head. Western site values were lower than central site values but overlapped, with western $\Delta+$ ranges from 80.1-83.4 and 79.5-82.6 at central sites. The eastern Breaker Bay and Palmer Head sites had lower than expected diversity. For MMI, eastern sites again were heterogeneous (86.3-87.6) but so were the central sites (87.8-88.0). Ranges were much smaller in this group. Western sites were very similar taxonomically.

3.5 Beta diversity

PERMDISP identified significant differences in MA composition ($F = 3.089$; $df = 9, 179$; $p(\text{perm}) = 0.006$). No clear spatial pattern emerged from the 13 significant pairwise comparisons out of 45 ($p < 0.05$, Appendix H). At the scale of "Area", β -diversity did not differ significantly ($F = 1.953$; $df = 2, 186$; $p(\text{perm}) = 0.195$), although non-significance between "West, East" was borderline ($p = 0.053$). This non-significant

trend can be seen in both the MDS plot (Figure 4.6, top row) and the average distance-to-centroid percentage values (Appendix H). In the MDS plot, eastern sites separated along the Y-axis, with western and central sites intermixed. Dissimilarity at the westernmost site (44.1%) was close to that of an easternmost site, Breaker Bay (37.9% dissimilarity), indicating a lack of linear spatial gradient in assemblage characteristics.

No significant differences in β -diversity were identified at the site level for MMI ($F = 0.856$; $df = 9, 66$; p (perm) = 0.742) but were at the level of "Area" ($F = 3.667$; $df = 2, 73$; p (perm) = 0.047). The pair "Central, East" differed ($p = 0.026$) with "Central" 39.2% dissimilar and "East" 44.6% dissimilar. The westernmost sites were most dissimilar (40.7%) and Breaker Bay the least at 29.6%. These two trends are less clear than the MA MDS plot but still discernible.

Species and higher taxa driving west-east trends were identified by the MDS vector overlay plots (Figure 4.6). For MA, eastern sites grouped together, with central and western sites intermixed. Four rhodophytes and *Macrocystis pyrifera* and *Carpophyllum* spp drove this pattern. *Caulerpa* spp (0.7% correlation), separated west/central from eastern sites. The invasive phaeophyte *Undaria pinnatifida* was not identified as significantly correlated to any area, indicating its very limited distribution in low abundances. For MMI, the sites were less clearly grouped by position along the coast and the dispersion of sites within each area was greater. The central/east difference in dispersion identified by PERMDISP can be seen along the Y-axis, with some overlap, with the western sites falling to one side of the data cloud. In this group, the brittle star *Pectinura maculata*, the sea star *Patiriella regularis*, the red crab *Plagusia chabris* and the two pua *Haliotis* spp contributed to the east/central split. Total numbers for the recreationally harvested blackfoot pua *Haliotis iris* increased from <1 at each western site to 4.5-11.8 at the central sites and 0.8-14 at the eastern sites. Numbers at Breaker Bay were the lowest of this group (0.8). The smaller size of the yellowfoot pua, *H. australis*, has protected it from recreational and commercial harvesting, but its total numbers followed a similar pattern. The sea urchin *Evechinus chloroticus* contributed to the apparent west vs. central/east trend. This species is also recreationally collected and its total numbers varied from 3.8-14.6 at the western sites to 1.8-2.3 at central sites to 1.8-11.1 at eastern sites. The gastropod *Turbo smaragdus* (0.7% correlation) contributed most to the overall multivariate structure.

Figure 4.1. Trends in univariate biodiversity indices (mean $\pm$ SE) along Wellington's south coast: species richness, number of individuals (macro-invertebrates)/percent cover (macro-algae), and Berger-Parker reciprocal values for two rocky-reef epibenthic groups. Different letters indicate significant differences in diversity indices between sites at $p = 0.05$ as identified by Tukey's HSD post-hoc tests. No letters indicate no significant differences between sites. Data were collected for two annual pre-reserve baseline surveys in 2008-09. $N = 8$ for all sites but Breaker Bay ($N = 4$).

Figure 4.2. Trends in univariate biodiversity indices along Wellington's south coast: Simpson's and Pielou's values for two rocky-reef epibenthic groups. Different letters indicate significant differences in diversity indices between sites at $p = 0.05$ as identified by Tukey's HSD post-hoc tests. No letters indicate no significant differences between sites. Data were collected for two annual pre-reserve baseline surveys in 2008-09. $N = 8$ for all sites except Breaker Bay ($N = 4$).

Figure 4.3. Spearman's correlations between two epibenthic groups for species richness (Simpson's index, $p = 0.57$) and community evenness (Pielou's index, $p = 0.95$). Data were collected from Wellington's south coast (2007-09).

Chapter 4. Local-scale biodiversity patterns in Wellington's epibenthic rocky-reef assemblages

Figure 4.5. Trends in species richness and equitability from rank abundance plots. B: Trends in average taxonomic distinctness for two epibenthic groups with funnel plots representing the 95% upper and lower confidence limits drawn from 999 permutations. Data were collected from Wellington, New Zealand's nearshore rocky-reefs (2008-09). Breaker Bay was only surveyed during 2009.

Macro-algal assemblages

Mobile macro-invertebrate assemblages

Figure 4.6. β -diversity patterns in two epibenthic groups from Wellington, New Zealand's south coast nearshore rocky reefs. Multi-dimensional scaling plots used Jaccard presence-absence matrices in PERMDISP to identify spatial patterns between baseline survey sites. Vector overlay plots show Spearman rank correlations of species with multivariate assemblage structure. * indicates correlations > 0.5%; ** > 0.7%. Data were collected between 2007-09.

4. Discussion

This study was the first of the baseline studies (Cole & Jackson, 1989; Pande, 2001) to characterise local and gradient biodiversity of two epibenthic groups from Wellington south coast's shallow rocky-reefs in terms of species richness, equitability, and taxonomic relatedness. Spatial patterns and trends in these community properties were identified, but the differences between sites were not always significant or unidirectional. No large shifts in diversity values that would indicate turnovers in assemblage composition were identified. For mobile macro-invertebrates (MMI), the westernmost sites beyond the Karori Rip at Sinclair Head were identified as species-poor and inequitable relative to the other sites. Central and eastern sites showed greater taxonomic variety and higher abundances. This pattern contrasted with the macro-algal group (MA), for which species richness and evenness were lower at the eastern end. The presence of spatial trends rather than significant differences along unidirectional linear gradients and the small average distances to centroids suggested that the scale of changes between western and eastern control sites versus reserve sites were small. Lack of a relationship between the two groups for richness and evenness indicated different responses to local conditions. The study also assessed the power of the baseline survey sampling effort to capture local diversity patterns and found that the effort was sufficient to capture local taxonomic variability of these two epibenthic groups in the rocky-reef depth stratum sampled.

At local scales (corresponding to the size of this study area), ecological diversity is driven primarily by historical and current interactions between and among biotic and abiotic variables, and secondarily by the size of the sampled areas and the intensity of sampling effort. It reflects the hierarchical nature of evolutionary diversity in time and space, with regional and larger-scale phylogenetic evolutionary processes structuring those at smaller scales (Gaston & Williams, 1996; Ray, 1991). Theories about larger-scale processes, discussed above and driven by temporal and spatial interactions between abiotic variables and the biological processes of speciation and extinction, do not apply to these micro- to meso-scale systems, where species responses to resource heterogeneity and physical and biological disturbances are primary factors. These biotic and abiotic processes and factors include resource competition (space, light,

nutrients), competition, predation, disease, settlement and recruitment, density-dependent and -independent mortality, energetics, wave energy, water chemistry, sediment, substratum type and availability, and anthropogenic effects (Dayton, 1985; Scheibling et al., 2009; Terlizzi & Schiel, 2009).

Partitioning out ecological sources of variation in local diversity is difficult for several reasons. First, knowledge of past events and processes, when available, is not always quantifiable for analyses or collected at comparable scales. Second, singling out specific variables or processes is complicated by the spatial autocorrelation of environmental variables and covariance of taxa. These interactions make testing the niche breadth theory, which postulates a hyperspace of required resources unique to each species, difficult. At best one can select several variables for experimentation but significant results will be qualified by possible correlations of untested variables (Benedetti-Cecchi, 2009). Third, the size of the sampled area can introduce sampling bias (Gaston & Williams, 1996; Ray, 1991). Fourth, the importance of factors will vary among taxonomic groups (Rosenzweig, 1995). Fifth, resource requirements can shift when a species' functional role changes to compensate for altered abundances of other species or groups (Kunin & Lawton, 1996).

The south coast kelp forest composition appears to have reached a degree of compositional stability based on the continuity of observed canopy species, of the mosaic distribution patterns of these and of mobile and sessile invertebrates, and of the taxonomic composition along the coast described in the two previous pre-reserve baseline studies and two stand-alone studies dating back over thirty years (Choat & Schiel, 1982; Cole & Jackson, 1989; Pande, 2001; Shears & Babcock, 2007). The dominance of the canopy laminarians *Ecklonia radiata* and *Lessonia variegata* and the furoid *Landsburgia quercifolia*, the presence of *Caulerpa brownii* beds, the lack of kina beds and paucity of herbivorous gastropods at a western/central site three decades ago (Choat & Schiel, 1982) resembled more recent descriptions of various south coast communities with only small differences in abundance and distribution (Cole & Jackson, 1989; Pande, 2001; Shears, N.T. & Babcock, 2007). Although most of these surveys targeted different sites along the south coast, with only Pande's sites overlapping with those of this study, my subjective assessment, based on a review of the abundance values reported in these studies, is that no major differences in

community descriptors occurred along the south coast between the 1982 report and this study that might signal community turnover.

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The correlation of *Caulerpa* spp with western and central sites is hard to explain given its tolerance of exposed conditions and the relative homogeneity of canopy composition and of south coast environmental conditions identified in chapter 2 (Adams, 1994). Again, no large shifts in any one taxon were observed that might create inhibitory effects to this genus' recruitment. Of interest is *Caulerpa* spp's inhibitory effect on other macro-algal functional groups and mobile macro-invertebrates. Only sessile invertebrates, primarily poriferans (and small fish) were observed to share space and this inhibitory effect continued after the alga had died: *Caulerpa* "graveyards" of sediment-filled stolon networks were barren. This biogenic structure appeared to inhibit recruitment, perhaps through an inhibitory effect on biofilms that condition substrata for subsequent recruitment by higher taxa (Laure et al., 2009).

The MA species richness patterns at the eastern end could be explained in part by resource theory, with habitat as the limiting resource elsewhere along the coast. The eastern sites Palmer Head and Flax Bay had the highest proportion of mobile substrata (boulder and smaller classes, chapter 2). The giant kelp *Macrocystis pyrifera* and four understorey Rhodophyta that contributed to spatial patterns in the β -

diversity MDS were present on these classes. The lower MA species richness, characterised by *Ecklonia radiata* stands, is difficult to interpret given the similarity of resources (habitat, physical parameters) along the coast. At all successional stages of kelp forests the effects of stochastic and deterministic processes affect taxa differently at different spatial scales and at varying stages of ontogenetic development. At more advanced stages, canopy density and plant size structure diversity through alteration of the physical habitat and by providing resource needs to other plants, sessile and mobile invertebrates and fish (Arkema et al., 2009; Benedetti-Cecchi et al., 2001; Creese, 1988; Davenport & Anderson, 2007; Estes et al., 2004; Irving & Connell, 2006a, 2006b; Jackson & Winant, 1983; Kennelly, 1987; Santelices & Ojeda, 1984).

Disturbance theory suggests that maximum diversity is attained when communities are subjected to intermediate levels of disturbance (Connell, 1979). Physical disturbance, to which Wellington's south coast (WSC) subtidal environment is subjected through surges, southerly swells, storms, tides, shifting sediments, strong winds (Carter, 2008) and biological disturbance in the form of grazing and canopy sweeping (Airoidi, 1998; Ayling, 1981; Kendrick et al., 1999) appear to maintain the WSC system in a state of non-equilibrium, thus supporting high diversity by clearing patches into which successional processes allow a range of species to compete. This on-going disturbance, although dynamic, leads to a compositional stability defined by the continued success of certain taxa. The biotic resistance hypothesis, suggests that such species-rich assemblages are better able to resist invasive species by possessing a greater diversity of functional groups that monopolise resources, leaving none available for invasives (McQuaid & Arenas, 2009). This hypothesis could well be tested in the near future by the presence of the Asian invasive laminarian *Undaria pinnatifida*. First introduced to New Zealand in ship's ballast water and found in Wellington Harbour in 1987, a local study found that it is highly adaptable to high energy and deeper sites than when first introduced. In southern New Zealand it spread 15km along an exposed coast in 7 years and competed with native canopy species, including *Lessonia variegata*, *Marginariella* spp, *Ecklonia radiata*, and *Lessonia quercifolia*, four species identified as significant along WSC (Hay & Luckens, 1987; Russell et al., 2008). A second southern study did not identify competition with native canopy but did detect opportunistic abilities to exploit open substrata (Schiel & Thompson, 2012). This

species' dispersal abilities, rate of expansion, and adaptability suggest that it might soon pose a threat locally.

A possible benefit of the apparent persistence of these two groups' composition might be resilience to species loss. Resilience can be considered from several theoretical viewpoints. The redundant species hypothesis, which states that only a subset of species is required to perform ecosystem services, suggests that a certain number of species could effectively disappear without changes in ecosystem status. A second projection, based on the premise that all species perform essential roles, is a decline in ecosystem goods and services. The idiosyncratic theory sees contributions as taxon-specific, with the outcome determined by which taxa are negatively affected. Ehrlich and Ehrlich's rivet (= species) hypothesis, suggests that currently not enough knowledge is available to identify which and how many species could be lost before an ecosystem would no longer function. The one functional group that all ecosystems clearly cannot afford to lose is that of the primary producers (Kunin & Lawton, 1996).

Knowledge of local-scale diversity patterns and their relationship to larger-scale patterns described in two national classifications of biogeography and coastal environmental zones (Beaumont et al., 2008; Shears et al., 2008) helps resource managers in two ways. Firstly, this work provides fine-scale descriptions that enhance the accuracy of these two national schemes and related themed management maps. Secondly, these results provide a basis for modelling system changes in ecosystem functioning related to future coastal activity and climate change, including rising sea levels and temperatures and increased ocean acidification.

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Chapter 5. Size frequency patterns as a measure of variability in Wellington south coast's rocky-reef assemblages

Patiriella regularis on *Ulva lactuca*, Yung Pen, Wellington south coast

1. Introduction

1.1 Using structural features of key species in community assessment

Temperate rocky-reef macro-algal forests rank as one of the most productive marine and terrestrial ecosystems, with annual primary productivity often exceeding $1,000 \text{ g carbon m}^{-2} \text{ year}^{-1}$ (Mann, 2000). Two genera of the order Laminariales (kelp) dominate rocky reefs worldwide: *Macrocystis* and *Ecklonia*. A third genus, *Laminaria*, is common to northern hemisphere temperate hard bottom communities, while *Lessonia* spp characterise southern hemisphere systems (Barnes & Hughes 1999). Associated with these genera is a range of Fucalean and other canopy and understory species that provide food and refuge to fish, sessile and mobile macro-invertebrates, including gastropods and echinoids. Macro-invertebrates structure benthic communities through processes that include predation, grazing, and competition for nutrients and space (Fletcher 1987; Valiela, 1995; Mayfield et al., 2001; Naylor & Gerring, 2001).

These three macro-algal genera have been described as engineer species because of their role in structuring epibenthic assemblages through bio-physical processes: they contribute nutrients and energy through photosynthesis; they provide nutrients and protection to fish and invertebrates as nurseries and as settlement habitat; and they physically modify assemblage structure and diversity through water motion-induced sweeping and shading (Norton & Benson, 1983; Eckman et al., 1989; Schiel, 1990; Anderson, 1994; Schiel, 2004; Wernberg et al., 2000; Vandendriessche et al., 2007). The subsidies of particulate and dissolved organic matter provided to macro-invertebrate grazers and filter-feeders in intertidal and deeper benthic systems from macro-algae can outweigh phytoplankton's contribution to those faunal functional groups (Duggins & Eckman, 1994; Bustamente et al., 1995; Britton-Simmons et al., 2009).

The mobile macro-invertebrates *Evechinus chloroticus* (sea urchin/kina) and the abalone species *Haliotis iris* and *H. australis* (blackfoot and yellowfoot paua respectively) are characteristic grazers of shallow subtidal reefs around New Zealand.

The urchin barrens created by kina grazing in large assemblages elsewhere in the north island and in the far south island have not been identified along Wellington's south coast (Cole & Jackson 1989; Schiel & Hickford 2001; Stewart & MacDiarmid 2003; Shears & Babcock 2007).

Identifying standing stock through models developed from morphometry and biomass can provide information that is useful in several ways. First, these patterns are a function of growth patterns, which in turn can reflect local biological and environmental conditions. Second, by identifying the scales at which structural features vary, resource management plans can be developed at the appropriate scale (Kelleher, 1999). Third, this information can be used to identify spatio-temporal patterns in productivity (see below). Finally, ecosystem functioning and productivity are two criteria essential to the development of biological valuation schemes that provide an integrated picture of an area's biological and ecological status for long-term spatial planning (Derosus et al., 2007)

Baseline information on primary productivity allows identification of and discrimination between short- and long-term changes due to anthropogenic coastal activity, and those due to natural variation. Primary productivity assessment methods include non-destructive techniques such as measurement of ambient nutrient levels in the water, chlorophyll *a*, dissolved oxygen levels, oxygen production, and ¹⁴carbon (Brinkhuis, 1985; De Wreede, 1985; Levine, 1985). Destructive harvesting of macro-algae and macro-invertebrates yields wet weight biomass and morphometric data that can be used to develop predictive equations for subsequent non-destructive assessment of standing stock. The data can also be used to develop energy conversion equations for use in the development of energy-flow models (Crisp, 1984; De Wreede, 1985).

Wet and dry weights are common metrics used for comparing abundances among taxa and among different habitats, ecosystems, and areas (Barnes & Hughes, 1999). They can also be used as a measure of energy resources available to higher trophic levels, and to evaluate human impacts on marine systems. Biomass also identifies baseline abundance levels for biological valuation management strategies (Derosus et al., 2007). For example, macro-invertebrate dry weight was used in the North Sea to assess the ecosystem effects of mineral extraction and trawling, as well as to identify their distribution gradients (Heip et al., 1992). Macro-algal biomass was

used in a large-scale decadal-long development of coastal bioregions off South Australia (Baker et al., 2008). Biomass has been incorporated into the European Union's Water Framework Directive monitoring protocol for identification of invasive macro-algal blooms (Scanlan et al., 2007).

Trophic web models have practical applications, among them ecosystem status and function assessment. For instance, biomass of macro-algae and sessile and mobile-macro-invertebrates was incorporated into a model for evaluating the El Niño-related sea temperature increases on altered benthic community abundances off Peru (Taylor et al., 2008). Trophic models can also be used to develop and evaluate the effectiveness of resource management choices: the impacts of the single-species fisheries management approach at 21 sites worldwide was assessed through modelling of trophic relationships (Pinnegar et al., 2000).

Because morphology is a plastic feature structured by local physical conditions (exposure, water storm- and surge-related movement) as well as biological pressures (predation, competition, disease, grazing), patterns in morphological variation can identify differences in physical processes at small and large scales (Gerard & Mann, 1979; De Wreede, 1985; Larkum, 1986; Thomas, 1986; Hurd, 2000; Mann, 2000). Incorporating phenological and spatial variability into sampling allows development of the most accurate predictive models for management and coastal development policy. Thus, there is a need for field techniques that allow rapid, accurate collection of structural information.

1.2 New Zealand studies on structural characteristics of benthic species

Several New Zealand studies have used structural features and standing stock to understand ecological responses to protection measures and the role of environmental variables on phenology and hybridization. Energy-flow models have been used as management tools: a recent study used a trophic structure model to assess the functioning of the Te Tapuwae o Rongokako Marine Reserve (Pinkerton et al., 2008) off the North Island's east coast. A model has also been developed for the Wellington south coast to assess the recovery of this area from long-term fisheries impacts following the creation of a marine reserve in August, 2008 (Eddy et al. submitted). Both studies identified a lack of available biomass data for many trophic

groups, including macro-algae and mobile macro-invertebrates. In a study on the development of a biogeographic classification scheme for nearshore rocky-reefs, Shears & Babcock (2007) used biomass to develop predictive regression equations for 33 red, green, and brown macro-algal species collected from the north and south islands (Wellington south coast excluded).

Macro-invertebrate biomass of four mobile species was used to identify their contribution to energy and material flux in a north island rocky reef system (Taylor, 1998), while Choat & Schiel (1982) used biomass as part of a large-scale study on subtidal rocky-reef community structure. Morphometrics were used to address hybridization and environmental response patterns within the *Carpophyllum* genus of the North Island's East Cape area (Hodge, 2009). In a local study along Wellington's south coast, morphometrics and biomass were used in a productivity study of a subtidal rocky-reef engineer species, *Lessonia variegata*. This study's biomass prediction equation provided a non-destructive tool to assess standing stock for commercial harvesting at times that minimise harvesting's negative impact (Schwarz et al., 2006).

1.3 Chapter aim

My aim in this chapter was to assess variation in assemblage structure using size as a metric. These data could then be used to develop predictive models of wet weight for a range of key epibenthic species that would provide easily obtainable non-destructive estimates of benthic standing crop from Wellington's south coast. Specifically, I aimed to identify sets of morphometric features that could be easily measured in the field and that could be used to estimate wet weight biomass for a suite of dominant canopy and understory macro-algal species. These equations will update a previously developed model for a canopy species, and update models for two macro-invertebrate species of recreational and commercial interest (sea urchin and abalone). The results of this study were also used with abundance data from the two annual baseline surveys (chapter 3) in a concurrent study (Eddy, 2011) to develop the first trophic models of Wellington south coast's nearshore community and its response to historic fishing pressure (Eddy et al., submitted).

A secondary aim was to identify spatial patterns in size variation that might indicate environmental conditions specific to the collection sites' location along Wellington's south coast. It should be noted that the partial random collection method, described below, must be considered when assessing size frequency and morphometric data and that these patterns might differ from those obtained from future, fully randomly collected samples.

This study presents the first set of location-specific predictive equations to Wellington's south coast for a group of key macro-algal and mobile macro-invertebrate species from sites representing the full extent of Wellington's south coast. It also presents the first assessment of location-specific morphometric variation in key species from Wellington's south coast.

2. Methods

2.1 Field methods

Species were selected for collection based on three criteria: 1) their status as characteristic New Zealand shallow rocky-reef species; 2) their recreational or commercial interest (macro-invertebrates) (Choat and Schiel, 1982; Cole and Jackson, 1989; Schiel & Breen, 1991; Cole et al., 2001; Schiel & Hickford, 2001; Shears & Babcock; 2007); or 3) having been identified as commonly occurring during preliminary dives along 25m transects at sites along Wellington's south coast for this study. These criteria identified twelve macro-algal species in four orders (Caulerpales, Dictyotales, Fucales, Laminariales) and three macro-invertebrate species in two classes (Echinoidea, Gastropoda). Macro-algae were: *Caulerpa brownii* (C. Agardh) Endlicher, *C. flexilis* Lamouroux, *Carpophyllum flexuosum* (Esper) Greville, *C. maschalocarpum* (Turner) Greville, *Cystophora scalaris* J. Agardh, *Ecklonia radiata* (C. Agardh) J. Agardh, *Landsburgia quercifolia* (Hook. F. et Harvey) Harvey, *Lessonia variegata* J. Agardh, *Macrocystis pyrifera* (Linnaeus) C. Agardh, *Marginariella boryana* (A. Richard) Tandy, *M. urvilliana* (A. Richard) Tandy, and *Zonaria turneriana* J. Agardh. Mobile macro-invertebrates were *Haliotis australis* Gmelin (yellowfoot paua), *H. iris* Gmelin (blackfoot paua), and *Evechinus chloroticus* Valenciennes (kina).

Collections were made during daylight in austral winters (June-September) and summers (December-March) between 2007 and 2009. To obtain samples that were

representative of Wellington's south coast (WSC) shallow-water rocky-reef system, three sites along its length were sampled: First Wash (-41.36034, 174.71162) (west end, boat site), Princess Bay (-41.34643, 174. 78822)(central, boat/shore site), and Breaker Bay/Palmer Head (-41.33292, 174.83086/-4134622, 174.82289) (east end; boat sites) (Figure 5.1). Two nearby sites in the east were required to accommodate undiveable sea conditions at this end during the two-year period. The depth strata sampled were 7-12m for macro-algae and 3-12m for macro-invertebrates. To avoid collecting in the same area in which two annual baseline surveys of the benthic community were made concomitantly (chapter 3), starting sites in each area were shifted at least 100m away from the starting waypoints used for the baseline study using GPS and GIS tools. To minimise the risk of non-independent collections due to resampling of previously harvested areas, within each collection area the starting point was shifted using GPS to measure distances from previous waypoints. For the shore site, different entries were used at the central site (Princess Bay) to eliminate transect cross-overs that would affect independence of samples.

Figure 5.1. Collection sites for samples used to assess patterns in size frequency for key canopy and understory macro-algae and mobile macro-invertebrates along Wellington's south coast. Collections were made from nearshore rocky reefs during austral winter and summers, 2007-09.

No collections were made from the central site during the summer seasons due to unavailability of sea tables for specimen storage (for sample sizes by factor level see Appendix I). Blackfoot paua (*Haliotis iris*) were only present in the western area during the first collection (Winter 2007) and no yellowfoot paua (*H. australis*) were observed

at that end of the coast over the two-year collection and baseline survey period. Permission for collection of undersized paua was obtained from the Department of Conservation.

Samples were selected to ensure inclusion of a full size range for each species (Levine, 1985). At each site, a transect heading was selected from a random-numbers table. If a reef ended before the full set of 20 samples had been collected, then a new random heading was selected. Samples were collected at intervals selected from a second random-numbers table and these intervals were measured by kick-stroke. Therefore, transect selection and initial locations of specimens were fully randomly determined. However, when specimen sizes along a transect occasionally represented only one end of the size range, a short distance perpendicular to the transect heading was swum to collect a specimen in the missing size class. The first specimen encountered that met this requirement was collected. This method deviated from a fully random method but was the quickest way to ensure inclusion of the full range of size classes present along WSC when dive time was limited.

Collections of the colonial *Caulerpa* spp included *C. brownie* and *C. flexilis* because the two were typically too intermingled within a single patch to sort and weigh separately. *Cystophora scalaris* and *Sargassum sinclairii* were only collected during the first year due to low abundance in collected areas despite their initial apparent abundance (this may have been due to my initial unfamiliarity with the area during the first year). *Marginariella* species were only assessed during the second collection year due to species misidentification during the first year. First-year samples that could have been misidentified were therefore discarded. Abundance of the giant kelp *Macrocystis pyrifera* was only high enough for collection at the eastern site, although a less abundant and smaller stand did exist at a very sheltered central site (Island Bay).

Holdfasts of *Carpophyllum* species, *Landsburgia quercifolia*, *Marginariella* species, and *Sargassum sinclairii* were difficult to separate from the substratum into which the haptera had often grown. Therefore, the stems of these species were cut flush with the holdfast so that total wet weights of these species do not include their holdfasts. Holdfasts of the remaining species were included in total plant wet and dry weights. The two genera that grow in colonial-type patches, *Caulerpa* and *Zonaria*

turneriana, were collected using 10cm x 10cm quadrats. To minimise the effect of spatial, light, and nutrient competition on plants, only one individual of *Carpophyllum* spp was collected from each stand/holdfast. *Macrocystis* was the only species to consistently lose blades during the collection, retrieval, and storage process, so that biomass estimates obtained represented an underestimate of actual values. However, only small numbers of detached blades (~<10) were noted. A preferred collection method would place mesh bags over the entire plant before removing from the substratum in order to retain all the foliage.

2.2 Sample processing

Specimens were stored in plastic bags on the boat or on shore to minimise desiccation, then stored in sea tables with running seawater pending processing. *Caulerpa* spp and *Macrocystis pyrifera* were processed within 24 hours of collection because their tissue decomposed more quickly than the other species'. These samples were drained and rolled in newspaper to absorb surface water before weighing and measuring. Total weight was measured to $\pm 1\text{g}$. Stem/stipe weight was measured $\pm 0.1\text{g}$. Stem diameter was measured to $\pm 0.1\text{cm}$. Remaining species were shaken or knocked against the floor to dislodge excess water, hung to drain, then rolled in newspaper to dry before weighing. This process did not appear to damage samples. Epiphytes and holdfast sediment were removed before weighing.

Paua were blotted with newspaper before measuring green weight. To identify biomass of living tissue only, flesh was then removed from its shell and the bag separated from the foot to obtain separate foot and visceral weights (Crisp, 1984). This minimised total flesh weight differences due to varying gut content. Paua were also sexed based on gonad colour. Kina were drained aboral end up for ten minutes before weighing and measuring. Paua shell length and kina test diameter were measured $\pm 5\text{mm}$. Accessing paua shell height for measuring to include as a possible predictor variable in regression was not possible due to paua's use of inaccessible crevices as daytime refuges.

Morphometric and meristic features selected for use as possible predictor variables in regression analyses (Table 5.1) were chosen for the ease and speed with which they could be measured in the field to replace future destructive biomass

sampling. This was important for two reasons: 1) WSC has a limited number of diveable days per year, making selection of easily measured features a priority; and 2) selected features needed to be easily identifiable and measurable by volunteer divers unfamiliar with macro-algal structures and underwater sampling work. Therefore, features and replicate measures which might be useful for more specific aims such as hybridisation, and reproductive studies were not necessarily selected for this study.

Numbers of primary branches were counted for *C. flexuosum* but not for *C. maschalocarpum* due to the difficulty of distinguishing between primary branches and the main stem of the latter species. The lack of variation in values of the variable “stipe width” for *Cystophora scalaris* made it a weak candidate for inclusion in a predictive regression model. The variable “number of blades” was not counted for *Macrocystis* because blades were often detached during collection, transport, and storage. When subsets of blades/lamina were measured for length and width, blades selected haphazardly and different sets measured for these features to avoid correlation. Length and width measured along the longest and widest axes, respectively. When multiple stipes were present (*Landsburgia quercifolia*, *Macrocystis pyrifera*), a measure of the “length to first branch/blade” was made along the thickest stipe (Appendices I-L: collection, biomass and morphometrics summaries).

Dry weights from the first season's specimens, winter 2007, were dried for 24 hours in a 60° oven until a constant weight was obtained (Levine 1985). Plants were either dried intact (*C. flexuosum*, *C. maschalocarpum*, *L. quercifolia*, *S. sinclairii*, *Caulerpa* spp, *Z. turneriana*) or in parts: holdfast, stem/stipe, blades/fronds (*E. radiata*, *L. variegata*, *M. pyrifera*, *Marginariella* spp). Before weighing, plants were held in a cooling chamber for a species-specific period of time determined by that day's room climate and by trial-and-error. This minimised rehydration.

2.3 Statistical analyses:

SPSS Statistics v18.0.0 was used to develop predictive regression models. The “linear regression” module was used rather than the general linear module option to access the user-specified method of entry. The stepwise entry method was selected to test the null hypothesis that all model parameters equalled zero and that there was no relationship between the dependent variable “wet weight” and the predictor variables used for each species (Quinn & Keough, 2006). This method of variable entry was selected because it assesses each factor's statistical contribution separately. This approach, despite the arguments against its use, suited the lack of inherent ranking in plant structures that might otherwise have suggested an entry order other than ease of measuring in the field (Tabachnik & Fidell, 2007). To ensure the robustness of this approach, results were compared with those obtained using hierarchical and forward entry methods. Concordance among results was uniformly noted. Squared semi-partial correlations, adjusted R^2 , and standardized beta coefficient values were used to identify the relative contribution of each independent variable. Unstandardized beta coefficients were used to develop predictive equations for plant wet biomass (Quinn & Keough, 2006). P-values were considered significant at $p < 0.05$.

To identify the contribution of the factors “Year” (2007, 2008), “Season” (Winter, Summer), and “Site” (west, central, east) to variations in wet weight, generalized linear models (GzLMs) were developed for each species with morphometric features as covariates. Distributions were specified as normal with “identity” as the link function. The GzLM approach was selected because it allows the inclusion of both discrete factors and continuous variables. Because samples and differences among slopes and intercepts for significant predictor variables were small, all data were pooled across these three factors for the modelling described above.

When additional plant features measured for a subset of the collection were assessed, they were entered in a hierarchical regression method after the full stepwise variables were entered as a block. This approach allowed assessment of their contribution to the model (Tabachnik & Fidell, 2007). In this instance, the models' adjusted R^2 values and the ease of measuring significant plant structures in the field were considered when selecting the final model because the Akaike information criterion is not available through SPSS' “linear regression” module. Small increases

(<2%) in adjusted R^2 values were not considered sufficient justification for incorporating an additional variable requiring measurement in the field. For sporophytes that were structurally complex, thus difficult to measure in the field, a hierarchical model was run using only the easily measured variables and the resulting regression equation presented as an alternative. The “enter” variable method was used for the three mobile macro-invertebrate species, which had only one predictor variable. General linear model (GLM) tests with “sex” as covariate were used to identify possible differences in regression models between male and female *H. australis* and *H. iris*.

Test assumptions of normality, linearity, and homoscedasticity were assessed with residuals scatterplots of independent and dependent data. Correlations between independent and dependent variables were assessed using Pearson's product-moment correlation coefficients. Collinearity among independent variables was assessed with Pearson correlation coefficients, tolerance and VIF values. Outliers were assessed visually and using Mahalanobis distances (Tabachnik & Fidell, 2007). Log_e transformation of all independent variables was sufficient for data to meet test assumptions so dependent variables were not transformed. Mean weights 0.01 m⁻² for *Caulerpa* spp and *Zonaria turneriana* are presented in Table 5.2.

To identify possible spatial trends in species morphometry, PRIMER's PERMDISP test was used. This method tested the null hypothesis of homogeneity of morphometry data set variation among sites (Anderson et al., 2008). Data were normalised and resemblances matrices developed on Euclidean distances for permutation testing (9999). P-values were derived from permutations.

Wet weight data were not analysed because of the quasi-random collection method. Possible spatio-temporal patterns were tentatively identified using summary plots.

3. Results

3.1 Principal aim: Predictive models for estimating standing stock

3.1.1 Macro-algae

Models for all species were significant (Table 5.2). These explained from 38% of the variation in wet weight for *Caulerpa* spp, to 82% for *Macrocystis pyrifera*. For macro-algal species except *Zonaria* and *Caulerpa*, alternative models using increasing numbers of predictor variables for increased predictive power are presented. These range from the simplest one-variable model using plant length to models incorporating up to four features (*L. quercifolia*, *M. pyrifera*). The two genera with the fewest structures suitable for field measurement, *Caulerpa* and *Zonaria*, shared the lowest adjusted R^2 values: 0.38 using the maximum length of *Caulerpa* filaments per quadrat ($n=20$) and 0.41 for the same feature for *Zonaria*. ($n=59$). For these two, mean weight per unit area provides a more robust estimator to use in modelling standing stock (*Caulerpa* 39.5g per quadrat ± 1.4 ; *Zonaria* =18.8g per quadrat ± 2.5). The other species for which only the feature "plant length" was regressed was *Cystophora scalaris* ($n=46$) and its coefficient of determination was also low: 0.60. Different structures were identified as significant in the models for the two congeners, *Carpophyllum* and *Marginariella*. Explanatory power of the models varied between species: the highest adjusted R^2 obtained for *C. flexuosum* was 0.72 compared with 0.63 for *C. maschalocarpum*. The best goodness of fit for *Marginariella boryana* was 0.82 but only 0.58 for *M. urvilliana*. The lowest goodness of fit for the remaining macro-algal species was 0.72 for *Ecklonia radiata* and the highest 0.82 for *Macrocystis pyrifera*.

3.1.2 Mobile macro-invertebrates

Goodness of fit was lowest for the sea urchin *Evechinus chloroticus* using test diameter as the predictor (0.63), but higher for both abalone species. "Sex" was not a significant predictor for *H. australis* green ($F = 0.033$, $df = 1$, $p = 0.856$, $N=81$) or foot weights ($F = 0.577$, $df = 1$, $p = 0.451$, $N=81$) but was for *H. iris* green weight ($F = 5.257$, $df = 1$, $p = 0.026$, $N = 114$), although the differences in values between intercepts (<0.2) and slopes (0.002) were minimal. "Sex" was not a significant predictor for *H. iris* foot weight ($F=0.331$, $df = 1$, $p = 0.568$, $N=114$). Adjusted r^2 values were higher for *H. iris* models of foot and green weights using shell length as the predictor variable (0.88, 0.94) than for *H. australis* (0.72, 0.78 respectively). Confidence intervals (Table 5.3) for all parameters were relatively narrow, indicating that biomass estimates derived using

these equations based on relatively small sample sizes would provide estimates close to the population mean (Fowler & Cohen, 1993).

3.2 Spatial patterns in plant size

Significant differences in morphometric variation along the south coast were identified using PERMDISP for four of the twelve macro-algal and two of the three macro-invertebrate species (Table 5.3). Spatio-temporal contributions to morphometric variation for both epibenthic groups were also confirmed by GzLM results (Figures 5.5, 5.6). Of the four macro-algal species (*Carpophyllum* spp, *Ecklonia radiata*, *Lessonia variegata*) for which morphometry varied significantly, differences for three (*C. flexuosum*, *E. radiata*, *L. variegata*) were between the two south coast boundary sites (West, East) and the central site ($p < 0.05$, see Table 5.3). Variation did not differ between the westernmost and easternmost sites for these three. This spatial pattern also applied to the abalone *Haliotis iris*. For one macro-algal species, *C. maschalocarpum*, and one macro-invertebrate species, the sea urchin *Evechinus chloroticus*, morphometries at the western site were identified as significantly different from the central and the eastern sites. Morphometry for these two species did not differ between the central and eastern sites.

Common to all 6 plots were overlapping distributions. This lack of clear grouping of morphometric features among sites indicates that variability of features from each site were fairly equally dispersed and similar in multivariate space. This result was seen clearly in *C. maschalocarpum* and *E. chloroticus*; for these species, only a few observations fell outside the groups. Differences in PERMDISP dispersion were limited to a few observations. Differences in dispersion of variance were more evident in plots for *C. flexuosum*, *E. radiata*, and *L. variegata*. These plots suggested that morphological variation was less at the central site and greater at the western and eastern sites. Points for the western *H. iris* separated slightly but these results could have been influenced by the lack of sampling replicates. Overall dispersion patterns suggested that differences in morphological variation for all species were of degrees rather than of absolute differences.

3.3 Preliminary observations on patterns in biomass

Visual summaries of wet weight by site, season, and year for macro-algal and macro-invertebrate species (Figures 5.3, 5.) were presented with the caution that these samples were not fully randomly collected, and that dry weight data were only obtained on samples collected from the first season (winter 2007). Interpretations given here are presented only to suggest possible patterns that need to be explored through fully randomly-obtained data.

The wet weight data was characterised by an apparent lack of large-scale fluctuations in mean wet weight and mean total length among sites and between seasons and years for all species (column A, Figure 5.2). A possible temporal trend between year/season was suggested for *Macrocystis pyrifera*, where wet weight seemed to be lowest for summer 2008. A second possible trend was suggested by lower mean wet weights at the central site for both *Carpophyllum* spp, but this interpretation is based only one collection season since none were observed during the second year at this site. *Lessonia variegata* wet weight was slightly lower at the central site, while *Caulerpa* spp wet weight was lower at the eastern end. *Zonaria's* wet weight appeared to be lower at the western end of the shore. The percent total wet weight by plant structure (holdfast, stem/stipe, blades/fronds/ laminae) (column B, Figure 5.2) appeared to be consistent across seasons, years, and sites.

Apparent similarity of plant structure dry weight contribution to whole plant dry weight at all sites was observed for *C. maschalocarpum*, *E. radiata*, and *L. variegata* (Figure 5.4). Spatial patterns for total plant dry weight corresponded to weight wet patterns for *C. flexuosum* but not for *C. maschalocarpum*. Dry weight for *Caulerpa* spp, *Z. turneriana*, *L. quercifolia*, and *L. variegata* resembled wet weight patterns across the three sites. Only *E. radiata's* mean plant dry weights, which appeared to increase from west to east, contrasted with whole plant wet weight patterns, which appeared to have similar values at each of the three sites.

Feature	Total L ¹	W ²	HF ³ L	HF W	Stem/stipe L: HF to 1st blade ⁴	Stem length ⁵	Stem/stipe width ⁶	Number of stipes/stems off holdfast ⁷	Number of branches ⁸	Girth of stipe bundle ⁹	Rib ¹⁰ width	Mean length: lamina/ fronds ¹¹	Number of blades/ fronds
Group/Species													
Macro-algae													
<i>Carpophyllum flexuosum</i>	✓		NA	NA	✓		✓		✓			✓	
<i>C. maschalocarpum</i>	✓		NA	NA	✓		✓		✓				
<i>Caulerpa</i> spp	✓ ¹²												
<i>Cystophora scalaris</i>	✓								✓				
<i>Ecklonia radiata</i>	✓		✓	✓	✓		✓				✓		
<i>Landsburgia quercifolia</i>	✓		NA	NA	✓		✓	✓	✓				
<i>Lessonia variegata</i>	✓		✓	✓	✓		✓						✓
<i>Macrocystis pyrifera</i>	✓		✓	✓	✓		✓	✓		✓		✓	
<i>Marginariella boryana</i>	✓		NA	NA	✓	✓	✓					✓	
<i>Marginariella urvilliana</i>	✓		NA	NA	✓	✓	✓						✓
<i>Sargassum sinclairii</i>	✓		NA	NA	✓		✓		✓				
<i>Zonaria turneriana</i>	✓ ¹²												
Macro-invertebrates													
<i>Evechinus chloroticus</i>		✓											
<i>Haliotis australis</i>	✓												
<i>Haliotis iris</i>	✓												

¹L (macro-algae) = plant length from bottom of holdfast (when included) to tip of longest frond/blade. ²W (macro-invertebrates) = length along longest axis; width across widest axis. ³HF = Holdfast. ⁴Measured from top of holdfast to lowest axil with a blade/frond. ⁵Length of stem from top of holdfast past first blade to stem apex. See Figure 5.x, *Marginariella* spp. ⁶Width measured at widest point above the holdfast. ⁷For *M. pyrifera* the total number of stipes was counted midway between the holdfast and canopy. ⁸Primary branches only were counted. ⁹Measuring tape snug but not taut around stipes. See Figure 5.x. ¹⁰Rib of lamina blade. See Figure 5.x, *E. radiata*. ¹¹Five lamina/fronds were selected haphazardly. ¹²Patch held upside down and length measured from top of stolon/HF to longest blade/filament.

Table 5.1. Morphometric and meristic features used in the development of predictive regression equations for wet-weight biomass of 11 macro-algal and 3 macro-invertebrate species from Wellington (New Zealand) south coast's shallow rocky-reef system. Collections were made seasonally in austral winter/summers over a two-year period from three representative sites: west end (1st Wash), central (Princess Bay), and east end (Palmer Head/Breaker Bay). See Appendix H for seasonal sample sizes by year.

Species/Group	Notes	Model	Adjusted R ²	F-statistic	df	p	N
See Table 5.1 for full list of (intercept with unstandardised beta coefficients, see part B for associated features entered. standard errors) ¹ PL = plant length. HF = holdfast.							
Macro-algae							
Inwet weight =							
<i>Carpophyllum flexuosum</i>	plant length only	= 3.174 + 0.018 * PL	0.69	405.76	1, 174	<0.005	176
	plant length included	= 2.631 + 0.018 * PL + 0.410 * stem width	0.72	226.11	2, 173	<0.005	
<i>C. maschalocarpum</i>	plant length only	= 2.740 + 0.022 * PL	0.46	51.87	1, 58	<0.005	59
	plant length included	= 2.37 + 0.019 * PL + 0.102 * number of branches	0.63	50.13	2, 57	<0.005	
<i>Caulerpa</i> spp (<i>brownii</i> , <i>flexilis</i>)	length (longest filament)	= 2.163 + 0.006 * maximum length	0.38	13.01	1, 18	0.002	20
	mean wet weight	39.5 g ± 1.4 per 1% metre ²	-	-	-	-	180
<i>Cystophora scalaris</i>	plant length only	= 0.36 + 1.05 * PL	0.60	68.43	1, 44	<0.005	46
<i>Ecklonia radiata</i>	plant length only	= 2.042 + 0.047 * PL	0.58	272.96	1, 198	<0.005	200
	plant length included	= 1.333 + 0.056 * PL + 0.083 * HF width - 0.045 * stipe length	0.72	167.31	3, 196	<0.005	
<i>Landsburgia quercifolia</i>	plant length only	= 2.367 + 0.031 * PL	0.63	131.19	1, 78	<0.005	79
	quick field version	= 2.520 + 0.026 * PL + 0.828 * stipe width - 0.061 * length to first branch	0.71	65.09	3, 76	<0.005	
	plant length included	= 2.142 + 0.027 * PL - 0.048 * length of first branch + 0.65 * stipe width + 0.247 * number of branches	0.79	73.98	4, 75	<0.005	
<i>Lessonia variegata</i>	plant length only	= 4.789 + 0.030 * PL	0.57	67.27	1, 198	<0.005	200
	no plant length	= 5.405 + 0.017 * stipe girth + 0.041 * HF width + 0.017 * stipe length	0.73	178.72	3, 196	<0.005	
	plant length included	= 4.691 + 0.013 * stipe girth + 0.016 * PL + 0.036 * HF width	0.80	271.73	3, 196	<0.005	

Table 5.2. Regression models for predicting biomass of key species (by abundance, recreational and/or commercial interest) from Wellington's south coast. Regressions were run on ln-transformed wet weight using a step-wise entry method. Samples were collected from nearshore rocky-reefs in 2007-08 except as noted. Additional equations are presented when a second regression using additional measures made on a subset of data yielded higher adjusted R² values than for models using only the variable "plant length". P was considered significant at p < 0.05. See Table 5.3 for parameter confidence intervals.

Species/Group	Notes	Model (intercept with unstandardised beta coefficients) ¹	Adjusted R ²	F-statistic	df	p ²	N
Macro-algae							
<i>Macrocystis pyrifera</i>							
	plant length only	Inwet weight = 4.698 + 0.003 * total length	0.70	216.622	1, 90	< 0.0001	92
	quick field version	Inwet weight = 3.915 - 0.028 * mean stipe length to first branch + 0.040 * holdfast width + 0.247 * number of stipes + 0.770 * stipe width	0.71	52.212*	4, 87	< 0.0001	
	plant length included	Inwet weight = 4.500 + 0.002 * total length + 0.206 * number of stipes - 0.034 * mean stipe length to first blade	0.82	141.78	3, 88	< 0.0001	
<i>Marginariella boryana</i> ²	plant length only	= 2.322 + 0.034 * PL	0.71	233.36	1, 93	< 0.005	96
	plant length included	= 1.626 + 0.027 * PL + 0.055 * number of fronds + 0.372 * stem width	0.82	144.52	3, 91	< 0.005	
<i>Marginariella urvillia</i> ²	plant length only	= 2.302 + 0.040 * PL	0.58	94.88	1, 66	< 0.005	68
	plant length included	= 2.032 + 0.022 * PL + 0.113 * number of fronds	0.66	65.77	2, 65	< 0.005	
<i>Sargassum sinclairii</i>	plant length only	= 2.799 + 0.22 * PL	0.68	60.26	1, 27	< 0.005	29
	plant length included	= 2.085 + 0.021 * PL + 0.173 * number of branches + 0.376 * stipe width	0.78	34.98	3, 25	< 0.005	
<i>Zonaria turneriana</i> ³	length (longest filament)	= 1.098 + 0.099 * PL	0.41	41.66	1, 58	< 0.005	30
	mean wet weight	18.8 g ± 2.5 per 1% metre m ²	-	-	-	-	
Macro-invertebrates							
<i>Evechinus chloroticus</i>	test diameter	Inwet weight = 3.191 * 0.028 * test width	0.63	278.277	1, 164	< 0.0005	166
<i>Haliotis australis</i>	shell length	Ingreen weight = 1.713 + 0.035 * shell length	0.78	280.875	1, 79	< 0.0005	81
	shell length	Infot weight = 1.207 + 0.032 * shell length	0.72	210.537	1, 79	< 0.0005	
<i>Haliotis iris</i>	Shell length: ♂	Ingreen weight = 2.582 + 0.024 * shell length	0.94	494.707	1, 34	< 0.0005	35
	Shell length: ♀	Ingreen weight = 2.386 + 0.026 * shell length	0.94	565.993	1, 76	< 0.0005	77
	shell length: both sexes	Infot weight = 1.793 + 0.025 * shell length	0.88	804.289	1, 111	< 0.0005	

¹ See Figure 5.2 for plant measuring methods. Collected only in ² 2008 and ³ 2007.

Table 5.2, final. Regression models for predicting biomass of key temperate austral nearshore rocky-reef species.

Table 5.3. Beta coefficient values from models in Table 5.2 with their 95% confidence intervals. These values are from regression models for predicting biomass of key shallow rocky-reef macro-algal and macro-invertebrate species from Wellington's south coast. Regressions were run on ln-transformed wet weight using a step-wise entry meth on samples collected in 2007-08.

Species	Variable	Coefficient	95% confidence interval	
Macro-algae			Lower	Upper
<i>Carpophyllum flexuosum</i>	Total length	0.018	0.016	0.019
	Stem diameter	0.410	0.199	0.622
<i>C. maschalocarpum</i>	Total length	0.019	0.014	0.025
	Number of primary branches	0.102	0.062	0.142
<i>Caulerpa</i> spp (<i>brownii</i> , <i>flexilis</i>)	Length of longest filament	0.006	0.003	0.010
<i>Cystophora scalaris</i>	Total length	1.051	0.795	1.307
<i>Ecklonia radiata</i>	Total length	0.047	0.042	0.053
	Total length	0.056	0.050	0.062
	Holdfast width	0.083	0.061	0.105
	Stipe length	-0.045	-0.058	-0.032
<i>Landsburgia quercifolia</i>	Total length	0.031	0.021	0.032
	Stipe diameter	0.828	0.228	1.072
	Length to first primary branch	-0.048	-0.080	-0.016
	Number of primary branches	0.247	0.155	0.338
<i>Lessonia variegata</i>	Total length	0.030	0.026	0.033
	Stipe girth	0.014	0.014	0.021
	Holdfast width	0.041	0.029	0.054
	Stipe length (mean of 3)	0.017	0.010	0.024
	Total length	0.016	0.013	0.019
	Stipe girth	0.013	0.01	0.016
	Holdfast width	0.036	0.026	0.047
<i>Macrocystis pyrifera</i>	Total length	0.003	0.002	0.003
	Number of stipes	0.247	0.167	0.328
	Holdfast width	0.041	0.021	0.059
	Stipe diameter	0.77	0.116	1.424
	Mean stipe length to first branch	-0.028	-0.055	-0.002
	Total length	0.002	0.002	0.002
	Number of stipes	0.206	0.148	0.264
<i>Marginariella boryana</i> ³	Mean stipe length to first branch	-0.034	-0.052	-0.013
	Total length	0.034	0.029	0.038
	Total length	0.027	0.023	0.031
	Number of fronds	0.055	0.036	0.074
	Stem width	0.32	0.148	0.597

Table 5.3, Beta coefficient values from models in Table 5.2 with their 95% confidence intervals, final.

Species	Variable	Coefficient	95% confidence interval	
Macro-algae			Lower	Upper
<i>Marginariella urvillia</i> ³	Total length	0.040	0.032	0.048
	Total length	0.022	0.010	0.033
	Number of fronds	0.113	0.056	0.169
<i>Sargassum sinclairii</i>	Total length	0.022	0.017	0.028
	Total length	0.021	0.016	0.026
	Number of primary branches	0.173	0.076	0.270
	Stipe diameter	0.376	0.052	0.699
<i>Zonaria turneriana</i> ⁴	Length of longest blade	0.010	0.007	0.013
Macro-invertebrates				
<i>Evechinus chloroticus</i>	Test diameter	0.028	0.024	0.031
<i>Haliotis australis</i>	Shell length: green weight	0.035	0.031	0.039
	Shell length: foot weight	0.032	0.027	0.036
<i>Haliotis iris</i>	Shell length: green weight, ♂	0.026	0.022	0.026
	Shell length: green weight, ♀	0.024	0.024	0.028
	Shell length: foot weight	0.025	0.023	0.026

Table 5.4. PERMDISP tests for homogeneity of dispersion of morphometric data for key Wellington south coast benthic species. Data were normalised and converted to Euclidean distances. Collections were made during austral winter/summers, 2007-09. P-values were considered significant at $p < 0.05$ and ¹were obtained from 9999 permutations.

Test: deviations from centroid			Pairwise Comparisons			Mean z-values (from group centroids)			
F	df	p ¹	Group	t	p ¹	Group	Mean	SE	N
Macro-algae									
<i>Carpophyllum flexuosum</i>			<i>sqrt</i>						
9.0633	2, 153	0.003	East, West	2.2431	0.078	West	24.241	1.528	60
			East, Central	2.9158	0.0291	Central	11.688	2.349	18
			West, Central	4.0805	0.0009	East	24.241	1.528	78
<i>C. maschalocarpum</i>			<i>none</i>						
11.622	2, 155	0.0002	East, West	4.4817	0.0004	West	28.759	2.492	39
			East, Central	0.7787	0.5202	Central	38.739	1.622	39
			West, Central	3.3563	0.0061	East	40.505	1.37	80
<i>Caulerpa</i> spp. (<i>brownii</i>, <i>flexilis</i>)									
0.1136	2, 177	0.9033	East, West	0.4602	0.6617	West	8.226	0.626	80
			East, Central	0.1927	0.8521	Central	8.459	0.766	40
			West, Central	0.2243	0.8334	East	8.972	0.746	80
<i>Ecklonia radiata</i>									
5.8691	2, 197	0.0294	East, West	0.1853	0.8334	West	29.459	1.624	80
			East, Central	3.2752	0.0125	Central	21.170	1.831	40
			West, Central	3.1424	0.0127	East	29.887	1.643	80
<i>Landsburgia quercifolia</i>									
1.9189	2, 126	0.3213	East, West	1.4228	0.259	West	31.264	1.855	60
			East, Central	0.9284	0.4871	Central	39.393	2.697	9
			West, Central	1.6517	0.2036	East	34.945	1.804	60
<i>Lessonia variegata</i>									
5.0525	2, 197	0.0468	East, West	0.3539	0.7902	West	28.188	1.541	79
			East, Central	2.9858	0.0200	Central	20.746	2.1386	40
			West, Central	2.8118	0.0297	East	28.982	1.625	81
<i>Marginariella boryana</i> (2008 only)									
0.2991	2, 93	0.8299	East, West	0.6235	0.6376	West	28.314	2.334	36
			East, Central	0.6326	0.6155	Central	28.023	2.492	20
			West, Central	0.0798	0.9465	East	30.432	2.383	40
<i>Marginariella urvilliana</i> (2008 only)									
2.9305	1, 67	0.2371	East, West	1.7119	0.2445	West	36.718	3.423	28
			East, Central	<i>None observed at central site</i>		Central	-	-	-
			West, Central			East	29.496	2.5931	40
<i>Zonaria turneriana</i> (2007 only)									
1.014	2, 57	0.5136	East, West	1.0464	0.4109	West	29.231	2.926	20
			East, Central	0.3673	0.7772	Central	33.887	3.354	20
			West, Central	1.3879	0.2943	East	35.703	3.632	20
Macro-invertebrates									
<i>Evechinus chloroticus</i>									
6.5631	2, 163	0.0103	East, West	3.3423	0.0032	West	11.741	0.927	60
			East, Central	0.5535	0.6630	Central	18.703	2.141	42
			West, Central	3.3096	0.0045	East	17.36	1.376	64
<i>Haliotis australis</i> (green weight)									
0.1953	1, 79	0.7343	East, Central	0.4419	0.7399	West	-	-	-
			-	<i>None observed at western site</i>		Central	5.7428	0.8081	41
			-			East	5.2350	0.8167	40
<i>Haliotis iris</i> (green weight)									
11.406	2, 111	0.0033	East, West	2.4385	0.0826	West	4.445	0.54	14
			East, Central	3.2377	0.0299	Central	10.32	1.151	60
			West, Central	4.4632	0.0012	East	16.500	1.577	40

Table 5.5. Variability in biomass of key macro-algal species identified by generalized linear model using the factors "Year", "Season" and "Site". Data were collected during austral winters/summers, 2007-09 at three sites along Wellington's south coast. GzLM models used a normal distribution with identity link. P was considered significant below < 0.05.

Group/Species	Factor	Wald χ^2	df ¹	p
Macro-algae				
<i>Carpophyllum flexuosum</i>	Year	1.053	1	0.305
	Season	1.458	1	0.227
	Area	4.152	2	0.125
<i>C. maschalocarpum</i>	Year	11.345		0.001
	Season	1.118		0.290
	Area	0.022		0.989
<i>Caulerpa</i> spp (<i>brownii</i> , <i>flexilis</i>) ²	Year	0.434		0.510
	Season	2.289		0.130
	Area	41.641		< 0.005
<i>Cystophora scalaris</i>	Area	0.558		0.756
<i>Ecklonia radiata</i>	Year	6.005		0.014
	Season	6.912		0.009
	Area	10.973		0.004
<i>Landsburgia quercifolia</i>	Year	4.585		0.032
	Season	0.491		0.483
	Area	8.383		0.015
<i>Lessonia variegata</i>	Year	17.553		<0.005
	Season	0.017		0.895
	Area	1.476		0.478
<i>Macrocystis pyrifera</i>	Year	3.803		0.051
	Season	0.032		0.858
<i>Marginariella boryana</i> ³	Season	1.351	1	0.245
	Area	1.346	1	0.510
<i>Marginariella urvillia</i> ³	Season	0.000	1	0.984
	Area	2.910	1	0.088
<i>Sargassum sinclairii</i>	Area	5.817		0.016
<i>Zonaria turneriana</i> ⁴	Area ⁵	4.088	2	0.129
	Area ⁶	4.184	2	0.123

¹ Same values for all species unless otherwise indicated. ² Tested for wet weight only. ³ No "year" factor because samples were only collected in 2008. ⁴ No "year" factor because collections only during 2007. ⁵ For "height". ⁶ For "wet weight".

Table 5.6. Variability in biomass of key mobile macro-invertebrate species identified by generalized linear model using the factors "Year", "Season" and "Site". Data were collected during austral winters/summers, 2007-09 at three sites along Wellington's south coast. GzLM models used a normal distribution with identity link. P was considered significant below < 0.05. "Green weight" includes shell and soft tissue; "Foot weight", soft tissue only.

Group/Species	Factor	Wald χ^2	df	p		
<i>Evechinus chloroticus</i>	Year	6.899	1	0.009		
	Site	4.194	2	0.123		
<i>Haliotis australis</i>		<i>Green weight</i>		<i>Foot weight</i>		
		Wald χ^2	p	Wald χ^2	p	df
	Year	0.668	0.414	0.663	0.415	1
	Site	11.063	0.001	0.453	0.501	1
	Sex	0.072	0.788	0.011	0.915	1
<i>Haliotis iris</i>	Year	31.883	<0.005	85.746	<0.005	1
	Site	45.749	<0.005	13.74	0.001	2
	Sex	0.928	0.335	3.816	0.051	1

Figure 5.2. Spatio-temporal patterns in wet weights of dominant (by abundance) macro-algal species from Wellington south coast's nearshore rocky reefs (7–12m). Column A: mean wet weight (g) and height (cm) $\pm$ 1 SE. Column B: percent composition by weight of plant structures. Data are presented by austral seasons, year (2007–09), and area.

Figure 5.2, page 2. Macro-algal wet weight patterns for dominant species.

Figure 5.2, final. Macro-algal wet weight patterns for dominant species.

Figure 5.3. Spatio-temporal patterns in wet weights of key mobile macro-invertebrate species from Wellington south coast’s nearshore rocky reefs. **Column A:** Mean wet weight (g) and length (cm) or diameter (mm) ± 1 SE. **Column B:** Mean wet weight of plant components. Data are presented by austral seasons, year (2007-09), and area sampled.

Figure 5.4. Macro-algal dry weights presented by (top) total plant dry weight ± 1 SE, and (bottom) by percent composition of plant structures for the four species for which structural data were obtained. Species omitted due to extremely small sample sizes: *Cystophora scalaris* and *Sargassum sinclairii*. Samples were collected during austral winter, 2007 from Wellington, New Zealand's south coast nearshore rocky reefs.

4. Discussion

This study developed a set of location-specific predictive equations for standing stock of commonly occurring macro-algal and macro-invertebrate species along the WSC. The variables selected for inclusion in these models can be quickly measured in the field and easily taught to volunteer divers unfamiliar with structural features of macro-algae and macro-invertebrates. The equations presented here were also used in combination with abundance estimates presented in chapter 3 to develop the first energy-flow models of Wellington south coast's nearshore ecosystem in a concomitant study (Eddy et al. submitted). This study also identified spatial patterns in these species' morphometry. Also presented were provisional summary plots of possible spatio-temporal patterns in mean wet and dry weights of these species. The patterns in morphometry identified through regression models and PERMDISP suggest that physical processes that influence these features vary at small scales within the WSC area and that species respond differently to these scales.

4.1 Relevance of regression models and morphometry to assemblage studies

Understanding basic ecological patterns of abundance, size, distribution, scale, physical processes and the scale(s) at which these aspects of community structure occur is fundamental to the development of protection and management plans for assemblages and areas of interest. Patterns in productivity, morphometry, and standing stock identify spatial and temporal scales in physical processes, which in turn define ecological boundaries. Size-frequency data can be used to set sustainable size limits for commercially and recreationally harvested species. They can also define baselines against which protection and management measures and anthropogenic changes in ecosystem status can be evaluated. Regression models can be used to assess productivity using non-destructive standing stock estimates that can be used in ecosystem-function models. Morphometric variation can identify the scale at which physical processes such as water flow, nutrient input, and temperature change, and biomass estimates can be used for comparisons among communities. Patterns in productivity and abundance using biomass, and patterns in physical processes using morphometry, provide information on an area's biological and ecological status.

Resource planners can incorporate this with information on other aspects of biological and ecological status to develop estimates of biological valuation for spatial planning and resource management (Vasquez and Guerra 1996, Hurd 2000, Pirker 2002, Derous et al. 2007, Li et al. 2009).

4.2 Local patterns

Regression models of wet or dry weight include morphometric structures as predictor variables. Morphometric structures, biomass (wet or dry) and plant/macro-invertebrate recruitment and presence are highly influenced by local physical processes such as water flow, temperature, nutrient input, sedimentation, storm and wave activity, herbivory, predation, competition for space and light, depth, habitat availability and proximity. These processes are highly scale dependent along nearshore rocky reefs (Denny, 1987; McShane et al., 1994b; McShane & Naylor, 1995; Cole et al., 2001; Duggins et al., 2003; Binzer & Middelboe, 2005). Morphological plasticity enhances organisms' survivorship by reducing mechanical damage and displacement (Hay, 1981; McShane et al., 1994b). However, variation in morphology has also been identified as a response to anthropogenic stressors (Warwick, 1993).

Comparisons of regression models from different New Zealand regions to assess the robustness of this study's models are of limited use: Choat & Schiel's (1982) models of several canopy species included here obtained much stronger goodness of fit by regressing against dry weight for R^2 values greater than 0.90. Shears & Babcock's (1987) study of key macro-algal species around New Zealand did not include local sites but their models used wet weights and obtained parameters that differed from this study's results. The only study of a local macro-algal canopy species, *Lessonia variegata*, developed a more powerful model with an R^2 of 0.93 using only "holdfast diameter" and "stipe girth" as predictor variables (Schwarz et al. 2006). The difference in models might be due to site quality (Island Bay), which is sheltered, or to environmental conditions specific to that sampling year. Models of local *H. iris* growth data included environmental information as explanatory variables (Naylor et al., 2006) which render comparisons of goodness of fit difficult. Russell (2004) modelled both *Haliotis* species population structure for WSC but these did not include predictive models. Her study did note the scarcity and smaller mean sizes of *H. australis* observed

elsewhere in New Zealand (McShane et al., 1994a). Comparing regression models developed from different sites, dates, and from fully randomly collected must acknowledge the effect of natural environmental variation on growth and other biological processes.

Models for macro-algal and macro-invertebrate data reflect local conditions at the time of collection, but given the cost of surveying and the limited diving on WSC this set of models can provide useful information for future surveying. The alternative models developed here provide quicker or slower assessment options in the field to suit available time. The high coefficients of determination obtained by Choat & Schiel's (1987) regressions against dry weights suggest that this form of biomass is preferred when modelling standing stock, but the cost, time, and physical resources needed to obtain dry weights might be prohibitive. These factors must be weighed against dry weight's superior value as a measure of actual plant tissue without the confounding component of water content, which varies with the physical processes identified above (De Wreede, 1985).

Seasonal patterns in growth and size have been identified for benthic macro-algal and macro-invertebrate species, including *Ecklonia*, *Macrocystis*, and *Haliotis* spp (Kirkman, 1984; Larkum, 1986; Pirker, 2002; Russell, 2004). However, higher growth and productivity do not always correspond to periods of highest temperatures or light levels, for example, *M. pyrifera* canopy aerial biomass was greatest in austral winter and lowest in summer (Pirker, 2002). Interannual differences in plant allocation of biomass explains the different morphometric values observed for *L. variegata* at Island Bay (Schwarz et al., 2006) and for *L. variegata*, *L. quercifolia*, and *E. radiatia* at Owhiro Bay (Choat & Schiel, 1982). Seasonal patterns in WSC *Haliotis* spp growth and spatial patterns in size were identified during a one-year study (Russell, 2004). This spatial trend in variation in size was identified by PERMDISP in this study. This and other spatial differences identified by PERMDISP for four canopy and two macro-invertebrate species might be a function of site-specific physical processes identified above: the western and eastern sites are more exposed to prevailing southerly storms and groundswells while the collection reefs at the central site are buffered by several reefs located further offshore.

The absence of both *Haliotis* spp at the westernmost site of this study on all but one of the sampling dates differs from the abundances Russell (2004) recorded. Although they were lowest at her western site, which was the same used in this study, during her year-long study, a persistent *H. iris* population was observed. Paua and kina recruitment are affected by ecological and biological factors that occur on both small and large scales: exposure, wave action, and presence of conspecific adults and each other (McShane & Naylor, 1995; Naylor & Gerring, 2001; Naylor & McShane, 2001). Models of larval transport along New Zealand's coast suggest that recruitment to local and near-shore reefs can be affected by high-energy water movement, but that local populations can be self-sustaining (Stephens et al., 2006).

Given the on-going presence of local paua populations along the majority of WSC, an equally exposed area subject to strong hydrological and meteorological forces (chapter 1), the absence of paua at the western end during these collection dates is curious. The baseline survey related to this study (chapter 3) identified low abundances at the western sites during the same two-year period. One possible factor limiting local population size might be the Karori rip that runs west along Sinclair Head at the western end of WSC. This strong persistent current might act as a natural barrier to larval transport and reseedling of the heavily-poached western populations from the east. A second possible factor might be the presence of higher winds in this area, also known as "the wind factory" due to local topography that channels winds in this area to speeds typically 1-2 times faster than at central and eastern sites. Surface winds might interact with the Karori rip to contribute to offshore larval transport (Stephens et al., 2006). Given their previously observed presence in these conditions, a more likely anthropogenic structuring influence might be the on-going poaching at this remote and sparsely-populated end reported by local residents (pers comm). Removal of mature paua could be a factor limiting this population's ability to act as a sink and increase numbers from within.

Chapter 6. Patterns in benthic habitat and association: mapping Wellington south coast's nearshore zone

Acknowledgements (and apologies) to Hiroshige and Hergê

"It is not down on any map; true places never are..."
Moby Dick

1. Introduction

1.1. Marine Mapping

1.1.1. Mapping's role in conservation, management, and research

Benthic mapping can provide information on environmental attributes that contribute to community structure, such as bedform, slope, depth, orientation, seabed composition, distribution, range, and condition. It can also be used to assess biotic composition, condition, and habitat use. These associations between substratum and biological assemblages comprise biotopes, ecological units that can form the basis of management strategies and reserve designs (Costello & Emblow, 2006). Environmental management uses information on seascape and biotope attributes, such as location, diversity, distribution, condition and proximity to anthropogenic activities, to monitor changes in ecosystem status, develop functional and biological value maps, track benthic and pelagic recruitment and stock distribution, identify and describe populations of conservation, recreational and commercial interest, and designate marine reserve locations. (Jacobi & Langevin, 1996; Davies et al., 2001; Kendall et al., 2004; Stevens & Connolly, 2005; Eyre & Maher, 2011; Kendall et al., 2011). Habitat mapping has also been used to assess reserve-related long-term community shifts in trophic structure and develop physical surrogates for predictive management models (Huang et al., 2012; Leleu et al., 2012).

Geographical information software (GIS) is a powerful management tool for archiving, analysing, sharing, and displaying spatially-referenced acoustic survey data and associated ground-truthing video/still images. Digital maps and summary products can be easily amended, updated, and queried to provide information at a range of spatial scales and hierarchical resolution to facilitate comparison among areas of interest (Wright, 2001; Valavanis, 2002). Output can be readily modified to suit the technical expertise of stakeholders (Anderson & Yianopoulos, 2001; Palumbi et al., 2003). Marine applications include near- and offshore resource and geomorphology assessment, monitoring habitat changes related to aquaculture, assessment of anthropogenic impact on freshwater and marine systems, oceanographic and marine geological surveys, identification of range and availability of habitat essential to life histories of key species; assessment of trawling/dredging habitat degradation; and surveying areas for special biological or ecological

significance (Bremen, 2002; Gauthier & Rose, 2002; Thrush & Dayton, 2002; Foster-Smith & Sotheran, 2007; Murtojärvi et al., 2007; Beaumont et al., 2008).

In New Zealand, coastal mapping projects have developed benthic habitat maps for management at the Motonui area (Mead & McComb, 2002), Tuingara to Blackhead Point (Funnell et al., 2005), Waiwakaiho, (Cole et al., 2000), Taranaki (Crofskey, 2007), and CROP Marine Reserve (Remy-Zéphir, 2007). The Tonga Island Marine Reserve mapping project developed guidelines for the use of acoustic techniques in soft-bottom seascapes (Thrush et al., 2003), while the Te Whanganui-a-Hei Marine Reserve study evaluated biotopes under reserve protection (Haggitt & Kelly, 2004; Hewitt et al., 2004). Risk assessments for the Kapiti and Wellington Harbour region used mapping (Robertson & Stevens, 2007a), and two acoustic surveys (side-scan, multi-beam) have produced a 4-class geological description of Wellington's south coast (Aaron & Lewis, 1993) and a seascape description of the then-proposed Taputeranga Marine Reserve (NIWA, 2005). The largest-scale mapping project developed biological and physical classification schemes for New Zealand's exclusive economic zone (Snelder et al., 2005).

Spatially-referenced information on seascape and biotope attributes can be used with spatial planning and integrated coastal zone management software such as MARXAN, PANDA, and CLUZ, to develop and evaluate conservation and management scenarios and predictive models. This group of software planning packages uses community structure, biodiversity, geophysical, habitat, cultural, economic, social, and legislative data to divide spatial areas of interest into user-defined planning units. Users weight such data and manipulate planning unit size to develop alternative protection and management scenarios (Cook & Auster, 2006; Riolo, 2007; Ball et al., 2009; Watts et al., 2010).

When mapping for resource management or to meet conservation targets, spatial scale is a critical factor in a survey design because the surveyed area must incorporate, if location is known, or estimate, when location is not known, habitat that supports essential ecosystem processes such as recruitment, larval transport, ontogenetic development, nesting, and migration (Intelmann & Cochrane, 2006; Lundblad et al., 2006). Benthic mapping can facilitate the placement of reserve boundaries to minimise or avoid the division of critical habitat (Kerr & Grace, 2005).

1.1.2. The role of drop cameras in benthic surveying

Still and video cameras support acoustic mapping by collecting images that can be used to ground-truth and classify backscatter data for use in the development of predictive models. Less frequently, drop-down video cameras are used in stand-alone applications for seascape and biotope exploration. The advantages of this less

equipment-intensive approach are rapid, affordable coverage of larger areas; coverage of shallow areas beyond the range of remote sensing; identification of species and physical habitat; and production of permanent archive of visual records. Drop-camera work can also cover depths at which diver deployment would be unsafe or prohibitively costly. Drawbacks include the difficulty of detecting and identifying cryptic flora and fauna such as hydrozoans, bryozoans, ascidians, poriferans, and turfing macro-algae and the difficulty of surveying in topographically complex areas (Holt & Sanderson, 2001).

In New Zealand, reports on stand-alone drop-camera work have been more difficult to source. Drop cameras were used to describe and quantify the Hawke's Bay subtidal environment (ASR Marine Consulting and Research, 2005), and to assess the relationship of a key canopy species, *Ecklonia radiata*, to its physical environment off Taranaki (Crofskey, 2007). Video and surveys, in combination with acoustic data, identified reserve-related changes in habitat use for Mimiwhangata Marine Park (Kerr & Grace, 2005) and in trophic structure for CROP Marine Reserve (Leleu et al., 2012).

1.1.3. Classifying drop camera output

Classification of marine communities has until recently been made according to biotic assemblages rather than geological characteristics (Gonzalez-Mirelis et al., 2009). It is not an absolute process but rather dictated by the map's goal (Penrose et al., 2005). Standardized hierarchical schemes are increasingly being used for classification at spatial scales ranging from global to local using hierarchical units such as provinces, biomes, geomorphic features, and biotopes (Roff & Taylor, 2000; Harris & Whiteway, 2009). New Zealand's Marine Environment Classification is a local example of a standardized national scheme (Snelder, T. et al., 2005). A standardized output assists management and data-sharing by ensuring that results are comparable across regions (Costello & Emblow, 2006). In such schemes the highest-order level has the greatest discriminating power, with lower levels sharing increasingly greater similarity. Higher-level attributes will not necessarily be present within all sub-regions (Roff, 2009). Hierarchical classification schemes identify relationships between physical habitats and associated assemblages to create biophysical datasets that describe key seascapes; these seascapes can then be used as surrogates for biodiversity (Post et al., 2006; Buhl-Mortensen et al., 2009; Last et al., 2010; Compton et al., 2012).

1.2 The study area

See Chapter 1, section 2 for the survey area description.

1.3. Chapter Aim

The goal of this survey was to describe and quantify the physical characteristics and biotic associations of Wellington south coast's nearshore benthos. Specifically, I investigated patterns in seabed structure in terms of composition, mobility, profile, texture, depth, aspect, slope, colonisation patterns, arrangement, and habitat use by benthic species. This information can provide resource managers with an overview of large-scale spatial patterns in benthic composition and distribution. This benchmark summary of the current state of nearshore community structure can serve as a benchmark for evaluating long-term anthropogenic and natural changes, such as habitat degradation and loss due to increased storms (IPCC, 2013). A secondary goal was to assess the impact of sampling intensity, defined by the total number of drop-camera waypoints, and method of image analysis, semi-quantitative video vs. quantitative still image visual assessment, on the quantification of physical habitat and biotic assemblage area and area.

2. Methods

2.1. Methods and Sampling Design

2.1.1. Survey area, equipment, and field methods

The survey was conducted along an 11.3 km stretch of the Wellington south coast (WSC) and covered approximately 2,192 hectares over depths from 1.5-46.9 metres between 2006 and 2008 (boundary points: northwest -41.35946, 174.70518; southwest -41.36813, 174.70513; northeast -41.3362, 174.83965, southeast -41.35340, 174.83993). The Deep Blue Pro submersible camera (Ocean Systems, Inc.) was used. The unit has a 3.6mm lens with fixed focus (1" - ∞), light sensitivity of 0.1 lux and 768 x 494 pixel, or 520 TV lines, resolution. Lighting was provided by two external high-intensity LED battery-powered Pelican torches. A 70-meter co-axial umbilical transmitted camera signals to the surface for real-time viewing and data recording. The video-marking GPS link unit did not function so recording the ship's navigational system and echosounder to the video unit was not possible. The GPS starting points, GPS error, time and depth of each camera drop had to be recorded manually.

The camera was centre-mounted at a height of 60cm in the frame, with the lens oriented perpendicular to the seabed to avoid distorting area (Brinkhuis, 1985). The stainless-steel quadrupod frame had a 25cm x 25cm square base with torches mounted on opposite corners to provide uniform coverage (*Inset*, Figure

6.1). The base of the frame was labelled with reference scales in 1cm increments to help with size and area estimation. The total weight of the unit averaged 17kg, with adjustments made to offset the effect of the bottom current on its stability. Solid-fill floats for buoyancy attached to the top of the frame offset its high centre of gravity. A frame-mounted, rather than freely drifting, camera was built to provide camera/cable protection in rocky reef areas and to standardise filming height and sampling effort. The frame size, although smaller and shorter than desired, was selected after a pilot trial suggested that very limited seabed visibility at the seabed was an environmental constant along the south coast.

With the vessel in neutral gear, the camera unit was set on the seabed at each planned waypoint long enough to record a clear image of the underlying biotope (Figure 6.1). The time for this varied because the time for suspended sediment to clear the field varied depending on surge intensity and turbidity. Once a clear image had been recorded with the camera at rest on the seabed, the frame was lifted 1-2m above the seabed, with altitude determining turbidity, and allowed to drift for an estimated distance of 10-20m, depending on seabed and surface currents. In practice, drift distances varied widely depending on tide, current, and wind. At two haphazard intervals along the drift path, the camera was lowered twice more to the seabed to acquire clear images unaffected by camera or cable shake. Coordinates and depth from the boat's echosounder were logged as the camera landed on the seabed. Depths were not corrected to datum because the maximum tidal range was small (approximately 1.7 meters; www.linz.co.nz) and because tidal estimates scaled to the surveyed area were not available.

2.1.2. Survey design

Waypoints were plotted first with the navigation software RayTech (Raymarine), then with the package VNS v9.2.2 (NobelTec). After an initial set of waypoints at 100-meter intervals ($n = 40$), the remaining 574 waypoints were plotted at 200-meter intervals for a final total of 620 waypoints between the Wellington harbour entrance and Sinclair Head (Figure 6.2). Waypoints were referenced to the WGS 1984 datum. A grid design was selected over a random sampling design because this allowed detection of spatial variations in physical habitat and associated community structure (White et al., 2007). Video from sixteen waypoints were rejected on the basis of quality (see below) for a final working total of 604 camera drops. This included 5 dived waypoints to supplement the less-sampled Sinclair Head area. Information on sampling distances used in stand-alone drop-camera surveys was difficult to source, so interval distance was based on the only New Zealand study available at the start of the project, which used 200m intervals (Cole et al., 2000). Inshore access was limited by surface

conditions, tide and weather that changed hourly and/or over the course of the day. Offshore working depth was limited to a maximum of 50 meters by the length of the camera cable.

Figure 6.1. The drop-camera deployment used to survey Wellington south coast's biotopes. sampling method At each planned waypoint ("X" under camera unit) and at two additional haphazardly selected points along the drift path (quadrats along yellow arrow) the camera unit was placed on the seafloor to stabilise the image during recording. The biotope along the drift path was also filmed with the unit 1-2 meters above the seafloor as the boat drifted. *Inset:* stainless-steel camera frame with scale markers along bottom cross-pieces.

Figure 6.2. Distribution of waypoints (n = 605) for a drop-camera video survey (2006-08) of Wellington's South Coast nearshore biotopes. Waypoints are presented by site, area, and season, for archival convenience only. These classifications do not correspond to benthic spatio-temporal patterns. The large green polygon marks the Taputeranga Marine Reserve. Yellow polygons in the overview indicate areas in which camera data recorded to DVD were lost. The two small green polygons identify trial areas sampled at 100m intervals before adopting the 200m grid intervals.

2.2. Post-Capture Video Treatment

2.2.1. Video processing

Video data were first recorded to DVD using a generic DVD burner before changing to the more reliable recording medium of DV tape using a Sony DCR-TRV25E. No standard protocols for video format were found in a literature search, so the final choice of file format was found through trial and error using PC options and, eventually, expert advice from NatColl (Wellington). Data were captured, reviewed for quality, and backed up daily. DVD files were recorded in .vob format and converted to .wmv format in Windows Movie Maker, while DV tape data were acquired as lossless .mov files using Final Cut Pro. Both formats were processed in Adobe Premiere Elements and rendered at a final resolution of 720 x 576 pixels in the lossless .mov format for highest quality files at manageable sizes. Video analyses were made on raw images with no quality adjustments.

During processing, video footage was edited into files ("clips") corresponding to individual camera drop/waypoint. Extraneous footage, such as deck time and water column footage, was deleted. Two times were calculated for each video clips "total clip time", measured from the time the camera unit entered the water to the time it left the seabed; and "seabed clip time", measured from the time the unit landed on the seabed until the time the camera left the seabed. "Clip time" is used here and refers refer to "seabed clip time". A title card containing metadata that included drop number, date, time of day, depth, coordinates and GPS accuracy when available, site and area was inserted at the start of each clip. Site and area labels were included for archival convenience and are not indicative of spatio-temporal patterns in seabed or biota.

2.2.2. Selection of video clip length and still images

Video analysis requires standardised video clip times, while still-frame analysis can be made on regularly or randomly selected frames taken along a tow path (Bullimore, 2001; Holt & Sanderson, 2001; White et al., 2007). Clip lengths varied due to changing sea and weather conditions throughout the survey. In order to identify a standard video clip length that would include the maximum number of camera drops, yet be long enough to allow the extraction of three still camera frames for quantitative analysis, mean seabed clip times were calculated. A clip length time of 20sec was thus established as the standard clip length that retained the greatest number of clips. To assess the validity of the physical and biological classifications made for 20sec clips, the results were compared with classifications made with longer 40sec clips, and again with those made from the full length of the seabed clips when these were longer than 40sec. These results were used as an

informal way to validate the continuous coverage maps developed from these video and still frame analyses (see below).

From each 20sec clip, three still images were selected for analysis at randomly-generated intervals. If the image at a specified frame was obscure, the next clear frame was selected. If an organism or substratum was unclear, a 10-second bracket around the frame was scanned for a clearer frame. Consecutive random numbers were rejected to avoid pseudo-replication from sampling at the same position. Contiguous plots were accepted. When the camera frame was askew so that the entire quadrat frame was not in the field of view, an area outside the frame equal to the missing area was included to maintain the same viewing area.

2.2.3. Image analyses

Video clips were viewed at least three times for habitats more complex than sand, gravel, and pebble, on which biota were seldom observed. Seabed and biota attribute lists were selected from other mapping studies only if they could be consistently evaluated (Ball et al., 2006; Beaumont et al., 2009; Ministry of Fisheries and Department of Conservation, 2008). Attributes found to influence community structure that were included were: depth, substratum orientation, seabed mobility, rugosity, sediment class, roughness (microhabitat), and patchiness (Menge & Olson, 1990; Glasby & Connell, 2001; Madden et al., 2008; Scheibling et al., 2009).

The quality of video clips was assessed separately for substratum and biota and assigned one of four possible scores:

- "Good": most features clear, distinct edges. Identification to specified level with certainty
- "Okay": features mostly clear with some fuzziness. Identification to specified level with certainty.
- "Borderline": features typically indistinct & blurred. Identification only to higher functional group/taxonomic levels. Possibility of misclassification.
- "Reject": cannot distinguish morphotype or functional group, phylum or below, or substratum.

Biological abundance was assessed as percent cover in two ways: semi-quantitatively using the SACFOR scale for video clips, and quantitatively using the image analysis software ImageJ for still frames (Hiscock, 1996; Davies et al., 2001). The SACFOR method visually classifies target species/groups into one of six abundance classes: superabundant, abundant, common, frequent, occasional, and rare, with corresponding numerical estimates of aerial coverage provided (Hiscock, 1996). These categorical classes could then be converted to quantitative data for use in data exploration and analyses. This visual analysis process was facilitated by

viewing still images through an acetate sheet attached to the monitor on which smaller units were drawn to help estimate areas more accurately (Holt & Sanderson, 2001).

Still frames were treated as photo-quadrats and analysed quantitatively using ImageJ software to directly calculate area (Rasband, 1997-2009). Percent cover of biota was measured in each of the three still images, averaged for that waypoint, and scaled to 1m². Seabed was classified and assigned the modal label when a frame class differed. In the quantitative analysis of still frames, ImageJ software was used to trace the area of biota, which provided an exact measurement of percent cover for subsequent analyses.

Biological classifications were defined by dominant canopy species, the only group that could be consistently identified. Whenever visible, abundances were also estimated for functional groups: canopy, understorey, turfing, and encrusting macro-algae; and sessile macro-invertebrates. Identification was made to species when possible. Canopy classes were based on Shears and Babcock's (2007) descriptions of WSC communities and labelled for the species with majority coverage. In practice this corresponded to ~>75% coverage, estimated visually from video clips and directly from still frames using ImageJ. The metric "percent cover" was used rather than individual counts for both sets of analyses because discrete counts could not be consistently or reliably obtained. The final list of physical and biological attributes and classes formed a hierarchical classification scheme reflecting local benthic communities (Table 6.1).

Seabed classification was classified hierarchically to two levels, the first "major class" level based on the five-class New Zealand coastal classification scheme (sand, gravel, cobble, boulder, reef) plus "pebble" from Wentworth's sediment classifications. This provided a nationally defined and standardized classification level to permit comparisons with other studies (Wentworth, 1922; Snelder et al., 2005; Ministry of Fisheries and Department of Conservation, 2008). The second "subclass" identification level allowed classifications specific to the seabed composition recorded along Wellington south coast (WSC).

Table 6.1. Final hierarchical classification scheme for the Wellington South Coast nearshore biotope mapping project. Labels and definitions for 1-5 taken from the Ministry of Fisheries & Department of Conservation's 2008 New Zealand coastal classification scheme.

SUBSTRATUM: Class and description		
<i>Levels 1-5 and descriptions directly from Ministry of Fisheries and Department of Conservation, 2008. Level 5A and modifiers adapted from various classifications presented in Ball et al., 2006. Size ranges for "Pebble" and "Cobble" defined by Wentworth, 1922.</i>		
LEVEL	LABEL	DESCRIPTION
Level 1	Biogeographic regions:	14 defined by unique combinations of mesoscale physico-chemical and oceanographic factors and processes.
	Wellington South Coast (WSC)	North Cook Strait <i>Transition zone between northern and southern flora and fauna (See Chapter 1 for detailed area description).</i>
Level 2	Environment	Two are defined.
	WSC	Marine
Level 3	Depth	Three depths zone defined overall.
	WSC	Shallow subtidal: mean low water spring – 30m depth. Deep subtidal: 30m – 300m depth.
Level 4	Exposure:	Defined by aspect, fetch, coastal openness, seabed profile, and wave climate.
	WSC	Low: Fetch < 50km; current ≤ 3 knots; e.g. sheltered bays.
		Medium: Fetch 50 – 500km; e.g. open bays.
Level 5	Habitat	Seven total based on sediment size classes & origin (organic vs. inorganic).
		Sand: 0.07-2mm.
		Gravel field 2-4mm.
		Cobble field 4-64mm.
		Boulder field 64-256mm.
		Rocky reef Basement platform.
Level 5A	Substratum	= (MFish/DoC "Habitat"). Classes below supplement those above.
Major class	Subclass	
Sand	Fine	Smooth appearance: Sand grain edges typically not distinguishable. Small shell pieces often present either intermixed or on ripple crests. Pebble, cobble occasionally present (≤5% of total area).
	Coarse	Sand-papery appearance: Grain edges typically distinct. Pebble or shells, intact or pieces can be present intermixed or along wave crests and/or troughs. Might include gravel. Pebbles ≤5% of total area if present.
Pebble field	Mixed sand/pebble	Sand >50% with pebble. Cobble and/or boulders occasionally present (≤5% of total area).
	Pebble field: 4-64mm	Typically continuous layer but occasionally the underlying sand matrix is visible in small (<5m diameter) patches. Cobble occasionally present (≤5% of total area).
Cobble field	Cobble in mixed pebble/sand Cobble: 64-256mm	Either continuous cobble or cobbles dominant with patches of pebble and/or sand/gravel in varying proportions. Boulders occasionally present; if so in ≤5% of total area.
Boulder field	Boulders in mixed cobble Boulders: >256mm	Boulders common to occasional, typically patchy, rarely continuous, in matrix of cobble and smaller classes present singly or mixed.
Patch reef	Patch reef in sand	Bedrock <10m diameter in sand fields. Occur singly or in few to many patches. Entire contour visible or inferable.
	Patch reef in mixed cobble	As above in fields of cobble and/or pebble/sand.

Reef	Reef/boulder with rubble	Bedrock dominant (>50% of total area) bordered by or littered with boulders and smaller classes. Sand patches occasionally present.	
	Reef	Bedrock. Surface broken, rough, with depressions, gutters, and/or channels that bisect reef to seabed with sand to boulder substratum.	
SUBSTRATUM: Class and description, continued			
5A		CLASS MODIFIER WITH DESCRIPTION	
5A.1	Arrangement	Unconsolidated, loose:	Sand, Pebble, Cobble, Boulders.
		Unconsolidated, packed:	Sediments embedded in matrix: pebble fields.
		Consolidated:	Reef.
5A.2	Relief	Ripples	: ≤10cm.
	Sand	Dunes	: >10cm.
5A.3	Texture	Shell hash:	Intact shells and larger (> 5mm) pieces.
		Smooth:	no visible surface irregularities.
		Rough:	irregular, pocked surfaces, often due to presence of crustose coralline algae.
5A.4	Height	Flat:	
		Low:	≤1m high: Large boulders, reef.
		High:	>1m high: Reef.
		Mixed:	Reefs.
5A.5	Structure	Gutters, depressions:	Surface grooves and concavities.
		Channels, depressions:	Channels bisect reef to seabed of sand to boulders; ≤3m wide; includes gutters.
5A.6	Profile	Horizontal:	
		Inclined:	Dominant orientation (> 50% of surface).
		Vertical:	
		Mixed:	No single orientation dominant.
5A.7	Distribution	Continuous:	Features regularly distributed.
		Patchy:	Features distributed in clumps.
5A.8	Cover	Colonised:	Epibenthic biota visible.
		Uncolonised:	No epibenthic biota visible.
5A.10	Seabed current	No effect:	No visible movement of plant or animal structures; at rest.
		Moderate effect:	Plant/animal parts move ~45% from rest position.
		Heavy effect:	Plant/animal parts move from still to horizontal.
5A.11	Seabed turbidity	None:	No particulate matter visible.
		Light:	Particulate matter visible but does not obscure features.
		Heavy:	Particulate matter dense, appears as snow. Tends to obscure features.
BIOLOGICAL CLASSES			
Level 6		BIOTA: MACRO-ALGAE*	
6.1	Abundance, distribution	Sparse:	Blades not touching, substratum clearly visible among plants.
		Medium:	Blades touching, substratum clearly visible among plants.

Dense:	Blades touching, substratum seldom or never visible.
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BIOLOGICAL CLASSES, continued

6.2	Dominant canopy species	<i>Ecklonia radiata</i>
	"Mixed" canopy: abundance dominated by the first species listed.	<i>Carpophyllum flexuosum</i> , <i>C. maschalocarpum</i>
		<i>Lessonia variegata</i>
		<i>Landsburgia quercifolia</i>
		<i>Macrocystis pyrifera</i> /other species: <i>E. radiata</i> , <i>C. flexuosum</i> /maschalocarpum
		<i>Marginariella</i> spp (<i>boryana</i> , <i>urvillia</i>)
		Mixed canopy: above species, two or more
6.3	Dominant understory functional groups	Red foliose, filamentous algae: <i>Euptilota formosissima</i> , <i>Gigartina circumcincta</i> , <i>Rhodymenia dictyota</i> , <i>Sarcodia flabellata</i>
	<i>This storey was not always visible so the recorded presence and abundance under-represented true values. Therefore these species lists represented a subset of the WSC nearshore benthos.</i>	Brown foliose, filamentous algae with brown algae: <i>Carpomitra costata</i> , <i>Halopteris</i> spp, <i>Sargassum sinclairii</i> , <i>Zonaria turneriana</i>
		Foliose green algae: <i>Ulva lactuca</i>
		Mixed understory: above and other unidentified species
6.4	Turfing groups	Geniculate coralline algae: 3 species, singly or mixed: <i>Arthrocardia</i> sp, <i>Cheilosporum</i> sp, <i>Jania</i> sp
		Foliose, filamentous red algae: <i>Anotrichium crinitum</i> , unidentified foliose red alga
6.5	Encrusting groups	Crustose coralline algae (CCA)
Level 7	BIOTA: FAUNA	
7.1	Sessile Invertebrates	Phylum Porifera: morphotypes (Boury-Esnault & Rützler, 1997) grouped
7.2		Class Anthozoa (Phylum Cnidaria): <i>Corynactis</i> sp, <i>Oulactis mucosa</i> , Unidentified deep species
7.3		Class Ascidiacea (Phylum Tunicata): solitary, social, compound
		Phylum Bryozoa: colonial
		Class Hydrozoa (phylum Cnidaria): colonial
7.4		Burrows: unidentified residents
7.5	Turfing, unknown	Flora or fauna; cannot be distinguished from video/still

Table 6.2. Semi-quantitative abundance classification scheme developed by the United Kingdom's Joint Nature Conservation Committee (Hiscock, 1996) for the visual analysis of biological assemblage video acquired for seafloor mapping projects.

MNCR SACFOR abundance scales							
S = Superabundant, A = Abundant, C = Common, F = Frequent, O = Occasional, R = Rare							
GROWTH FORM			SIZE OF INDIVIDUALS / COLONIES				
% COVER	CRUST / MEADOW	MASSIVE / TURF	<1 cm	1-3 cm	3-15 cm	>15 cm	DENSITY
>80%	S		S				$>1 / 0.0001 \text{ m}^2$ (1x1 cm) $>10,000 / \text{m}^2$
40-79%	A	S	A	S			$1-9 / 0.001 \text{ m}^2$ (3.16x3.16 cm) $1000-9999 / \text{m}^2$
20-39%	C	A	C	A	S		$1-9 / 0.01 \text{ m}^2$ (10x10 cm) $100-999 / \text{m}^2$
10-19%	F	C	F	C	A	S	$1-9 / 0.1 \text{ m}^2$ $10-99 / \text{m}^2$
5-9%	O	F	O	F	C	A	$1-9 / \text{m}^2$
1-5% or density	R	O	R	O	F	C	$1-9 / 10 \text{ m}^2$ (3.16x3.16 m)
<1% or density		R		R	O	F	$1-9 / 100 \text{ m}^2$ (10x10 m)
					R	O	$1-9 / 1000 \text{ m}^2$ (31.6x31.6 m)
						R	$>1 / 10,000 \text{ m}^2$ (100x100 m) $<1 / 1000 \text{ m}^2$

2.3. Statistical analyses

The PASW Statistics v18 (SPSS) package was used for χ^2 analyses, the software package "R", and the Excel macro add-on QI was used to conduct two-way analyses of variation without replication (Stephens, 2004). P-values for all tests were set at the conventional $\alpha = 0.05$ level. Pearson chi-square tests of association were used to test for associations between sampling intensity (number of waypoints) and seabed composition and biota, (Gotelli & Ellison, 2004). The areas of physical habitat and biotic cover derived from drop-camera waypoints were also tested for differences due to sampling intensity (number of waypoints) using two-way ANOVAs without replication. The four levels of the factor "sampling intensity" were the full set (605 drops), 450 drops, 300 drops, 150 drops. These sampling levels were selected arbitrarily. To randomly select the subsets of camera drops for each level from the full pool, ten draws of the appropriate number of randomly selected waypoints were generated and compared to the full, "true" frequency pattern. This was defined as the frequency distribution identified by the complete set of camera drops and comparisons among frequencies were made using the χ^2 test.

Additionally, the two pilot areas initially sampled at 100m intervals were compared to the same area when sampled at 200m intervals. This selection of waypoints was made by deleting every other waypoint in those pilot areas to obtain the 200m interval. To identify a possible effect of using quantitative versus semi-quantitative abundance assessments, the areas derived from each method were also compared.

Geostatistical analyses were run on quantitative macro-algal canopy percent cover derived from still-frame analyses using ArcGIS 10.1's Spatial Statistics toolbox. This field of analytical techniques, using georeferenced data, can be used to explore the spatial contribution to environmental and biological patterns, processes and interactions (Haining, 2003; Kanevski, 2008). A continuous bathymetry surface was generated from point depth data using ordinary kriging with a semivariogram model and cell size set to the shortest width/height of the point feature layer extent/250. This grid layer was used to generate a measure of the compass direction of WSC seabed, or aspect. It was also used to develop a measure of WSC nearshore slope. These features were selected because of their role in structuring physical processes and associated benthic assemblages (Bekkby et al., 2009; Costello et al. 2010).

Spatial statistics calculated for each canopy category included spatial autocorrelation, The Global Moran's I tested the null hypothesis of no spatial

autocorrelation and identified whether distributions were random, clustered, or dispersed. To measure central tendency and dispersion, the directional distribution calculated ellipses defined by standard deviation values. To measure hot spots for each canopy species, locations where abundance values deviated significantly from the norm, the Getis-Ord G_i^* statistic was calculated. To avoid skewing the analyses, the habitat "sand" was deleted from the dataset because no canopy were observed on this substratum (Allen, 2013).

Finally, patterns in habitat associations using Manly's selection ratio with the R package "adehabitatHS" This compares the known proportion of available habitat to the known proportion of habitat used by each species/group to calculate a ratio. Ratio values equal to 1 indicate random habitat use, larger values indicate positive habitat associations, and lower values indicate negative associations (Manly et al., 2002; Calenge, 2006). This test was run at two levels of identification: the more general classification level of functional groups comprising the visible epibenthos, and again for mixed species/functional groups to identify which species contributed to functional group patterns of habitat use. This approach might also allow comparison with other studies in which identifications were made to higher levels of resolution. The functional group list contained five levels: macro-algal canopy and understorey, mixed turfing (algae and invertebrates), sessile invertebrates (Ascidia, Bryozoa, Hydrozoa, Porifera mixed), and an unidentified deep anemone. The species list contained six dominant macro-algal furoid and kelp canopy species (Figure 6.4), understorey (red algae), understorey (*Caulerpa* spp), and understorey, other; turfing group, sessile invertebrates, and the deep anemone species.

Continuous coverage maps were developed using ArcInfo v10 (ESRI). Theissen polygons were developed from point/categorical data (waypoints), and then clipped to a boundary derived from the spatial extent of these waypoints (minimum convex hull clipped to the coastline). Polygons were then dissolved by attribute (substratum, biota) to create continuous-coverage maps of both components of Wellington south coast's habitat (Ormsby et al., 2009). Waypoints were reprojected into the New Zealand Transverse Mercator coordinate system for area calculations. A4 maps were developed at a scale of 1:50,000.

3. Results

3.1. Visual interpretation of patterns in benthic composition and distribution

The overall video quality was high, allowing use of the majority of the footage. Image quality scores were "good" or "okay" for 84% of both the biota and seabed

clips. Still-frame quality was rated “good” or “okay” for 92% of both biota and seabed clips. Classifications across the three sets of video clips (20sec, 40sec, and full) were consistent. Ninety-six percent (583) of canopy and seabed classifications assigned to the 40sec clips matched those for the 20sec clips. The differences were attributed only to identification of an additional group or species rather than a change in canopy or seabed class.

The representativeness of shorter video clips was very high; effectively the same information on biotic and abiotic components was obtained when compared with the two longer sets of video clips (40sec and full clip). When 20sec classifications were compared against those for full-length clips, 91% of the full clips’ canopy and substratum classifications matched. Of the 9% (53 clips) that differed, only 3% (17) were due to substratum being reclassified. The remaining 6% (35) of the clips differed only by the identification of an additional biological group or species that did not affect the primary canopy classification. The concordance of classifications suggests that viewing short extracts of longer video clips can provide representative information on biotic and abiotic patterns.

The Wellington south coast seabed was characterized as a heterogeneous mix of patchily distributed habitat dominated by mobile substrata consisting of sand and pebble fields offshore and at the eastern end of the coast. These distribution patterns were found both at the broader class level (Map 6.1) and at a finer level of resolution (Map 6.2). Larger, immobile substrata were more abundant from the central area westwards but overall, “reef/patch reef” comprised only 32% of the surveyed area. “Reef” and “patch reef” formed mosaics separated by gullies and channels and abutting large fields of small-class sediments. These two classes were also characterized by rough texture and broken, channelled surfaces. Rippled sand was confined to the eastern Lyall Bay area, while coarse sand dunes were found close to the harbour entrance. An extensive pebble field was identified south of a decommissioned quarry in the central area. The slope of the seabed was shallow along the WSC, with a few steeper surfaces located inshore (Map 6.5). The seabed aspect was dominated by a southerly exposure (Map 6.6).

A visual comparison of biological distribution maps developed from video and still-frame analyses (Maps 6.3, 6.4) shows that both techniques identified similar distribution patterns: anemone fields in deeper fields of mobile substrata, sessile invertebrate patches located in deeper areas, greater abundance of the canopy alga *Lessonia variegata* at the western end, algal canopy dominated by *Ecklonia radiata* along the length of the coast, and the restriction of the giant kelp *Macrocystis pyrifera* to small central and eastern patches. Differences between the two are the

video detection of a burrow field in Lyall Bay and south of Moa Point and a deeper sessile invertebrate patch at ~40m off Moa Point.

Directional distribution plots (Map 6.7) of standard deviations (1 SD ellipses) reveal subtle spatial differences. *Carpophyllum* spp, partially obscured by ellipses, has a western component, as does *Lessonia variegata*. *Marginariella* spp has a narrower distribution and *Macrocystis pyrifera* is limited to the central and eastern areas. *Ecklonia radiata* is the most widely distributed species and, with *L. variegata*, found at deeper depths relative to the other canopy species.

Hot-spot analyses, the identification of areas with significantly higher (or lower) abundances, identified similar patterns for *Carpophyllum* spp and *E. radiata* (Maps 6.8-9). High values for both species were widely distributed, although *E. radiata* values were lower off the Quarry and Red Rocks areas. *L. variegata* hot spots (Map 6.10) were limited to western and central areas while *Marginariella* spp hot spots (Map 6.11) were found primarily around the central Princess/Island Bay sites. As suggested by the directional distribution ellipse, *M. pyrifera* high values spots (Map 6.12) were, not surprisingly, limited to its distributional range at the eastern end.

Map 6.1. Seabed composition of Wellington south coast's narshore zone defined by major physical classes. Classes were identified by visual analysis of 20sec video clips acquired using a drop-camera deployed 1-2m above the seabed while drifting. The seabed was recorded at 604 waypoints between 2006-2008 for a total coverage of 2,193 hectares at 1.5-46.9 metres.

Map 6.2. Seabed composition of Wellington south coast's nearshore zone defined by sediment subclasses . Information was collected by stand-alone drop-camera video recorded 1-2m above the seabed while drifting. Subclass assignment was based on the relative percent cover of constituent sediment classes observed in each clip.

Map 6.3. Community composition of Wellington south coast's nearshore epibenthos defined by dominant taxa/groups identified from analysis of **video clips**. Drop-camera video clips were analysed for semi-quantitative abundance of canopy, understorey, turfing and encrusting groups, and sessile macro-invertebrates. A total of 2,193 hectares was surveyed between 2006-08

Map 6.4. Community composition of Wellington south coast's nearshore communities based on quantitative visual analysis of **still frames** extracted from video recordings. The taxa and functional groups used to classify habitats were those that were consistently visible in all video clips in which biota were present

Map 6.5. Aspect of Wellington south coast's nearshore seabed derived from a bathymetry grid developed from point depth data using a kriging function. Aspect identifies the proportion of seabed, or physical habitat, exposed to dominant oceanographic influences.

Map 6.6. Slope, measured in degrees, of Wellington south coast's nearshore seabed. Slope was calculated from a bathymetry grid developed from point depth data using a kriging function. Through slope's interactions with local and larger-scale oceanographic processes it contributes to biological patterns . Wellington's nearshore zone is dominated by plains, with steeper habitat located close to shore..

Map 6.7. Patterns in directional distribution defined by standard deviations (1 SD) presented for macro-algal canopy classes identified by quantitative analysis of stil frames extracted from a drop-camera video survey of Wellington south coast's nearshore zone. Patterns identify a *Macrocytis pyrifera* population locus at the eastern end of the shore, with distributions of the remaining species showing slight differences in range and by depth.

Map 6.8. Hot-spot analysis of *Carpophyllum* spp, mixed canopy abundance. The Getis-Ord Gi* statistics identifies the location of extreme values. Here, the pattern of high and low standard deviation values suggests a somewhat clustered distributions. Higher deviations from mean abundances (from percent cover) are shallower, as expected for photosynthetic taxa. Percent cover was calculated from still frames using visual image analysis software.

Map 6.9. Hot-spot analysis of *Ecklonia radiata* monospecific and mixed canopy abundance. The Getis-Ord Gi* statistics identifies the location of extreme values. Here, the pattern of high and low standard deviation values suggests a clustered distribution with higher abundance values based on percent cover at shallower depths. This hot-spot distribution closely resembles that of *Carpophyllum* spp. Percent cover was calculated from still frames using visual image analysis software.

Map 6.10. Hot-spot analysis of *Lessonia variegata* abundance. The Getis-Ord Gi* statistics identifies the location of extreme values. Here, the pattern of high and low standard deviation values suggests a spatial gradient, with higher abundances occurring centrally and in the western area. Percent cover was calculated from still-frames using visual image analysis software.

Map 6.11. Hot-spot analysis of *Marginariella* spp. The Getis-Ord Gi* statistics identifies the location of extreme values. Here, the pattern of high and low standard deviation values (1 SD) suggests a strong spatial gradient similar to that of *Lessonia variegata*, with higher abundances occurring centrally and in the western area. Percent cover was calculated from still-frames using visual image analysis software.

Map 6.12. Hot-spot analysis of *Macrocyctis pyrifera* abundance. The Getis-Ord Gi* statistics identifies the location of extreme values. Here, the pattern of high and low standard deviation values suggests a strong spatial gradient similar to that of *Lessonia variegata*, with higher abundances occurring centrally and in the western area. Percent cover was calculated from still-frames using visual image analysis software.

3.2. Statistical results

For the pilot area sampled at 100m intervals, the areas for each seabed class generated by Thiessen polygons were compared with areas generated by Thiessen polygons developed from the same waypoint data set that was resampled at 200m intervals. No differences in the areas of these seabed classes were detected ($p = 0.750$, Table 6.5). However, the 100m sampling grid did detect an additional seabed class, "boulder" that the larger-interval sampling did not identify.

Sampling intensity, defined as the number of camera drops, did not have a significant effect on the detection of seabed class or epibiota at either general or refined classification levels when compared for each image analysis method (video or still-frame) (all $p > 0.05$, see Table 6.3). This suggests that future mapping surveys could replicate this work using far fewer waypoints. However, the methods of visual analysis, video vs. still frame, did have an effect on output. The frequencies of seabed class and of epibiota did differ significantly (all $p < 0.014$, Table 6.4). In contrast, no significant difference was found for area ($p > 0.2$, Table 6.4). This discrepancy in significance might be driven by smaller areas (Figure 6.3).

Physical habitat use assessed at two hierarchical levels of biological classification ("major class" and "subclass") identified two patterns. First, habitat association did differ significantly among functional groups and specific taxa (Manly's selection ratio: $\text{Kni2L1} = 235.709$, $\text{df} = 44$, $p < 0.01$). Second, and a logical consequence of the first result, physical habitat was not being used in proportion to its availability levels (Manly's selection ratio: $\text{Kni2L1} - \text{Khi2L2} = 540.693$, $\text{df} = 4$, $p < 0.001$). Mobile and bedrock substrata were preferred, with "Reef", "patch reef", "boulder", and "cobble" assigned positive selection ratio values. "Pebble" was not selected (Figure 6.4A). A slight trend was present, with "patch reef" and "cobble" slightly less preferred than "reef" and "boulder". Of the functional groups driving these associations, turfing taxa showed the strongest association, with "boulder", followed by understorey and canopy, using "reef". Canopy also preferred "boulder". "Patch reef" was most associated with sessile macro-invertebrates. Only an unidentified anemone located in deeper areas (~40m) preferred the pebble habitat.

The preferences of the giant kelp *Macrocystis pyrifera* for the mobile substrata "cobble" and "pebble" was clear (Figure 6.4C). *Ecklonia radiata* shows the smallest fluctuation in preferences, which might explain its dominance along the south coast. The group of understorey Rhodophyta (Figure 6.4D) used "cobble" over other substrata, which might explain its abundance in the eastern Breaker Bay area. *Caulerpa* spp's preference for reefs corresponds to dived baseline survey

observations, which noted the presence of extensive stands of this understorey species on reefs at the central and eastern sites.

Spatial autocorrelation, a measurement of the degree to which a set of observations are related, or clustered) was identified for three canopy species ($p < 0.000$) (Table 6.3). *Carpophyllum* spp, *Ecklonia radiata*, and *Macrocystis pyrifera* were all classified as having clustered distributions, which was suggested by the epibenthic coverage maps (Maps 6.3-4), the directional distribution map (Map 6.5), and the hot-spot analyses (Maps 6.6-7 & 6.11). *Lessonia variegata* and *Marginariella* spp distributions did not differ significantly from random ($p > 0.42$ and 0.48 respectively). These different patterns suggest different responses to biological and environmental processes among canopy species.

Figure 6.3. Comparison of areas of biological classes identified by two methods of visual image analysis: video and still-frames extracted from video. Areas were generated by Thiessen polygons defined by semi-quantitative analysis for video clips, and fully quantitative visual analysis still frames extracted from clips. The total area of each benthic group generated from Thiessen polygons did not differ significantly between the two methods of visual analysis ($p > 0.05$). However, the frequency of camera drops that identified each benthic group did differ significantly between the two visual analysis methods (all $p < 0.05$).

Table 6.3. Spatial autocorrelation of macro-algal canopy classes calculated using the Global Moran's I test. Analyses were run separately for each species on datasets from which "sand" had been excluded to avoid skewing the results. No canopy species were observed on sand.

Canopy type	Moran's Index	z-score	p	Distribution
<i>Carpophyllum</i> spp, mixed	0.4513	18.6763	<0.000	clustered
<i>Ecklonia radiata</i> , monospecific & mixed	0.1818	7.6882	<0.000	clustered
<i>Lessonia variegata</i>	0.1563	0.804	0.42	random
<i>Macrocytis pyrifera</i>	0.1609	8.3217	<0.000	clustered
<i>Marginariella</i> spp	0.0126	0.6946	0.48	random

Table 6.4. Chi² tests of association between drop-camera sampling intensity and seabed and benthic composition. Seabed was assessed at two hierarchical levels of description: major classes and subclasses. Biological assemblages were assessed at two hierarchical levels of identification: functional group and the more specific level of canopy/group. The drop-camera video survey of Wellington's south coast nearshore zone was conducted between 2006-08. P was considered significant at < 0.05.

DIFFERENCES IN FREQUENCY OF SEABED CLASSES AND EPIBIOTA IDENTIFIED BY DIFFERENT SAMPLING INTENSITIES ASSESSED FOR EACH IMAGE ANALYSIS METHOD (video vs still frame)

Sampling intensities: Complete camera drop set (605) vs 450 vs 300 vs 150

		Physical habitat			
			Value	df	p-value
Seabed: major classes	Video	Pearson Chi-Square	7.24	15	0.951
	Still frames	Pearson Chi-Square	9.42	15	0.855
Seabed: subclasses	Video	Pearson Chi-Square	8.83	27	1.000
	Still frames	Pearson Chi-Square	12.24	27	0.993
N = 9605					
		Epibiota			
Community (15 groups)	Video N = 4375	Pearson Chi-Square	15.02	21	0.822
	Still frames N = 3748	Pearson Chi-Square	5.49	27	1.000
Functional group (3)	Video N = 4375	Pearson Chi-Square	6.02	6	0.421
	Still frames N = 3748	Pearson Chi-Square	1.97	6	0.922

DIFFERENCES IN FREQUENCY OF SEABED CLASSES AND EPIBIOTA WHEN COMPARED BETWEEN TWO METHODS OF IMAGE ANALYSIS. COMPARISONS WERE MADE ON COMPLETE SET OF CAMERA DROPS (N = 605)

Image analysis method: Video vs still frame

		Physical habitat			
Seabed: major classes		Pearson Chi-Square	14.27	5	0.014
Seabed: subclasses		Pearson Chi-Square	40.21	9	0.000
N = 1210					
		Epibiota			
Community (15 groups)		Pearson Chi-Square	5830.36	15	0.000
Functional group (3)		Pearson Chi-Square	157.90	2	0.000
N = 8123					

Test of effect of	Source of Variation	SS	df	MS	F	P-Value	F crit
100m grid vs. 200m for 2 pilot areas	West						
	Rows	177.237	5	35.447	20.643	0.002	5.050
	Waypoint interval	0.195	1	0.195	0.114	0.750	6.608
	Error	8.586	5	1.717			
East	Rows	453.153	4	113.288	71.435	0.001	6.388
	Waypoint interval	0.011	1	0.011	0.007	0.936	7.709
	Error	6.344	4	1.586			
Area of seabed derived from different numbers of camera drops	Rows	4297893.285	5	859578.657	2607.812	0.000	2.901
	Sampling intensity	148.608	3	49.536	0.150	0.928	3.287
	Error	4944.252	15	329.617			
Video vs stills on area of epibiota	Rows	5177946	8	647243.25	312.8836	0.000	3.438101
	Image analysis met	3901.38889	1	3901.3889	1.885969	0.207	5.317655
	Error	16549.1111	8	2068.6389			

Table 6.5. Differences in classes of seabed area derived from polygons generated by using different numbers of waypoints (all 605, 450, 300, or 150) or image analysis (video vs. still-frames). Differences were tested for using two-way analyses of variance without replication. Theissen polygons were dissolved across the relevant attribute (seabed or epibiota classification) for each test. P was considered significant at < 0.05.

Figure 6.4. Habitat associations between epibiota and seabed classes identified by a drop-camera survey of Wellington south coast's nearshore zone. Positive associations are indicated by positive Manly's alpha values (y-axes), negative associations by negative Manly's alpha values, and neutral association by the dashed line in each plot. **A** presents overall benthic association for each major seabed class; **B** examines associations by major functional benthic group while **C** & **D** examine habitat use at a finer resolution: **C** describes use by macro-algal canopy species and **D** for specific functional groups.

4. Discussion

In this study I described the marine landscape of Wellington south coast's (WSC) nearshore zone. I quantified the spatial distribution and quantity of seabed classes, or physical habitat at two levels of resolution, and summarized the seascape and a set of its attributes in continuous-coverage maps. These maps identified the highly heterogeneous and predominantly mobile nature of the seabed along WSC relative to the limited amount of solid bedrock and these results matched the seabed descriptions generated from the dived surveys of chapter 3. I related these patterns to the overlying biological communities and identified clear patterns of substratum use that differ among species and functional groups. Spatial patterns in distribution and abundance values of macro-algal canopy species were also identified.

Sampling intensity, defined by the number of camera drops, was shown to have little effect on the areas of seabed classes generated during the development of continuous coverage benthic maps. The medium used for visual analyses, video or extracted stills, was identified as having the potential to miss smaller areas. These results demonstrate that an effective stand-alone drop-camera mapping project can be completed without the oversampling of this project and with a very limited number of camera drops. This has implication for surveyors working in exposed areas where field conditions can make camera deployment difficult and greatly increase the risk of equipment damage or loss. Additionally, the general congruence of physical and biological information and spatial patterns identified by semi-quantitative and quantitative image analysis techniques is useful information because of the time-consuming nature of quantitative analyses using image processing software. This work has demonstrated that the much faster SACFOR analysis of short video clips produces accurate estimates of ecological patterns relative to fully quantitative methods.

This project, completed without access to technical expertise on mapping and navigation equipment, sampling design, visual analysis methods, classification techniques, or GIS support, operated with the bare minimum of equipment and this limited the spatial inferences that could be made from observed patterns. The inability to link GPS tracks to video was a factor limiting the development of more reliable continuous coverage GIS layers. Typically, mapping projects that use drop cameras are conducted as ground-truthing exercises to complement acoustic surveys and provide georeferenced data sets of physical habitat and associated biota that are incorporated into supervised classifications of acoustic signals. Because of the fine scale at which acoustic data is collected, this process permits the development of finely resolved raster-based continuous coverage benthic maps

(Davies et al., 2001; Penrose et al. 2005). Continuous coverage maps are easier to interpret visually than point data symbolized by category, but this type of coverage implies a link between the continuous layer categories and the reality they symbolize (Robinson, 1995).

In contrast, Thiessen polygons are generated from points using algorithms that do not rely on attributes associated with the points or other spatial information as a basis for the polygons created. They are mathematical structures, so not driven by ecological information. The attempt to ground-truth the polygons by comparing classifications made on shorter, 20sec extracts of the video clips with those made on longer, 40sec segments and full-length clips was based on the assumption that the longer the video clip, the more seabed was recorded, thus the more likely the categorized area of the Thiessen polygons would match the benthic characteristics they were intended to symbolize. At best, Thiessen polygons are a poor substitute for full coverage maps developed from mapping projects integrating acoustic with ground-truth data. As such, they were presented here to facilitate the visualisation of patterns in biotic and abiotic components of WSC benthos but it should be noted that their boundaries were not calculated from ecological data and are at best approximations of spatial patterns.

Categorization of these two components can be highly variable and subjective without a well-defined classification scheme, analysis protocols, or expert knowledge (MESH Consortium, 2004). New Zealand's five-level coastal classification scheme provided a useful starting point for classifying the seabed, but categorizing mixed-class substrata proved challenging, even after developing a personal rules-based system. Available literature presents a bewildering array of biological and physical classification schemes in which modifying attributes from one hierarchy become classes in another (Kerr & Grace, 2005; Ball et al., 2006; Robertson & Stevens, 2007; Snelder et al., 2007; White et al., 2007; Dauvin et al., 2008; Luceer and Pederson, 2008; Last et al., 2010). Shears and Babcock's (2007) rocky-reef community descriptors, used to develop biogeographic provinces of New Zealand's rocky reefs, provided a useful starting point for the classification of biota, but did not always capture the nuances of mixed canopy composition. Inexperienced mappers can find it difficult to recognize the ecological significance of mixed canopies of varying proportions and what the threshold of inclusion in categorisation should be.

Because modifiers such as rugosity, orientation, and texture, have the potential to affect the interpretation of acoustic data (Davies et al., 2001; Penrose et al., 2005), their status was recorded for each video clip and camera drop on the chance that the data set would be used to ground truth an acoustic survey conducted shortly before this project began. These data and the classification of

physical habitat to broader and finer levels of resolution would hopefully then allow acoustic mappers to select the level of detail most appropriate for the classification of backscatter data.

Stand-alone drop camera mapping techniques were extremely difficult to source and were only found after the project was started. The technique of boat drift from fixed waypoints in drop-camera surveys that was used here yields greatest accuracy in non-towed surveying (Holt & Sanderson, 2001). It has been used elsewhere in New Zealand for similar conservation-based mapping projects (Hewitt et al 2004, Kerr & Grace 2005). The technique of haphazardly selected sites around a fixed starting waypoint was also used by Kerr and Grace in their 2005 Mimiwhangata Marine Park (NZ) mapping project. Mean daily camera drop rate made in this project is in line with the 10-25 daily deployments identified as typical by the United Kingdom's Joint Nature Conservation Council field protocols (2001). Factors affecting this rate include weather, working depth, current, slack water duration, site topography, duration of power supply, technical malfunctions, and travel time (Holt & Sanderson, 2001).

Patterns in habitat association provide useful information to managers that they can use to enact measure protecting habitats that support species/functional groups of interest. There is a relatively limited amount of hard-bottom substratum along WSC and its aspect is primarily southerly, the direction from which strong and damaging storms and surges come (Carter, 2008). This orientation makes the nearshore zone vulnerable to the threat of more powerful and more frequent storms. Benthic habitat maps can serve as benchmarks for tracking changes in the condition of critical habitat due to extreme weather events. Additionally, long-term monitoring at the scale of decades using mapping and other survey techniques tracked track reserve-related changes observed at a northern New Zealand marine reserve (Leleu et al., 2012) and overall ecosystem status (Cogan et al., 2009; McArthur et al., 2010). These map products can be used to track changes in condition, cover, and habitat related to the Taputeranga Marine Reserve. The relationships between biotic components and abiotic variables can also be incorporated into predictive habitat models elsewhere (Stevens & Connolly, 2004; Orpin & Kostylev, 2006).

Hot-spot analysis, a type of clustering analysis (Anselin et al., 2006), provides another view on ecological processes that contributes to the benchmark picture of WSC rocky-reef ecological relationships. The differences in loci of high abundance values for canopy species, patterns which were also seen in the subtle differences of the standard deviation ellipses, identifies the taxon/group specific responses to environmental processes and biological interactions (Connell, 2005;

Wernberg et al., 2005; Tian et al. 2009). These associations and distributions can be used in future spatial planning.

The environmental patterns and habitat associations are relevant to resource managers because they identify the status of the ecosystem recently placed under reserve and allow structural change in WSC rocky-reef assemblages to be tracked in relation to the unprotected areas of the south coast (Stevens & Connolly, 2005). Biophysical data sets permit long-term tracking of anthropogenic changes to substratum and species presence, distribution, and abundance. In fact, short-term (days) and long-term (decadal) shifts in sediment cover have been identified by two previous acoustic surveys (Carter, 2008). Coastal planners can also use this data in decision-making software to evaluate alternative protection scenarios and possible amendments to local reserves (Ball et al., 2009).

One drawback to remotely collected data is its limited ability to capture patterns of cryptic, rarer, and smaller taxa (Davies, 2001). Drop-camera surveys can only record the emergent, larger, and taller taxa, leading to underassessment of understorey, turfing, encrusting, and sessile invertebrate species, with a resulting bias in biodiversity estimates. These drawbacks limit the representativeness of information derived from drop-camera video in structurally complex, high profile habitats. Although limited by depth and field conditions, dived surveys can access recessed habitats and detect taxa that are easily missed by or inaccessible to remote viewing methods.

Chapter 7. General Summary

The objective of this project was to investigate spatial and interannual variability in WSC's shallow rocky-reef epibenthic communities; identify the species/groups driving these patterns; the contributions of environmental variables to community structure; and the condition, availability and patterns of physical habitat use. These questions were examined at two spatial scales, fine and broad, using direct and remote survey methods.

Landscape ecology seeks to explain spatial heterogeneity in ecosystems through the study of structure, composition, ecological processes and sustainability of landscapes. Ecological patterns and processes are analysed within a spatial framework to investigate interactions between human activity and ecological processes (Pittman et al. 2011). This approach provides a conceptual framework for assessing the scales at which ecological processes operate in heterogeneous environments and has been increasingly applied to questions in coastal ecology (Bostrom et al., 2011; Pittman et al., 2011). Coastal benthic taxa have been found to respond differently to resource patchiness and changes so understanding the factors driving these differences allows managers to preserve beneficial conditions to maintain ecosystem integrity and goods and services. Impacts on resource availability have the potential to disrupt benthic ecosystem functions such as primary and secondary productivity and energy flow (Bostrom et al. 2011). In order to protect nearshore benthic processes, ecological patterns and the scales at which they operate for various taxa must first be identified. This study provides preliminary information on the ecological structure of Wellington south coast's (WSC) benthic communities and their relationships to a suite of environmental variables.

Dived and remote video surveys provide information at fine and broad scales respectively. Dived surveys require minimal equipment but are constrained by safety and training requirements and, often, adverse field conditions. Remote video surveys can be conducted in more adverse conditions but are costlier because of the equipment, personnel, software and training required. Data collected from diver surveys can be more immediately accessible conceptually to students with limited experience, contrasted with the effort and technical skills required to process and analyse remotely collected data. Software and hardware used in remote surveys require frequent servicing, updating and replacing, and are prone to malfunctions that can be costly in terms of time, money, and resources. Data collected from diver surveys can usually be processed quickly to produce ecological assessments.

A recent development is the increasing availability of more affordable remotely operated underwater vehicles for benthic resource assessment. In an era of international research networks, such equipment can often be made available to financially constrained research groups (Pauly & Clerck, 2010). Two points worth noting: first, it is not feasible to expect that every benthic researcher will have access to, time for, or interest in remotely operated vehicles, the software and hardware to run them, or the training and on-going technical support that will inevitably be required to address electronics issues. Electronics gear breaks down. Not all research programs can support the cost of ongoing support. Second, despite the physical constraints on dived surveys, divers can assess cryptic, hidden, and rare species that are typically obscured by canopy or complex substrata. Divers can enter shallow habitats where boats might not be able to venture and can also venture into structurally complex habitats that might snag remotely deployed equipment. To date, the dived survey remains a valid tool for a wide range of researchers requiring cost-effective, low-resource surveys.

The choice of survey protocols, whether dived or remote, is challenging when there is no nationally established database of baseline studies and where much of the national literature does not get archived in a timely manner. The choice to double the width of the belt transects deployed for the dived baseline surveys (chapter 3) was appropriate but the preferred choice would have been to increase the width to 10m x 25m, in keeping with several New Zealand baseline survey methods (Stewart & MacDiarmid, 2003; Haggitt & Kelly, 2004). Visual surveys of the rock lobster *Jasus edwardsii* clearly underestimated WSC populations in light of the commercial fishery this area supports.

The first pre-reserve baseline survey (Pande, 2000), targeted four mobile macro-invertebrate species on the basis of their interest to recreational and commercial harvesters, despite the abalone *Haliotis australis* not being a collection target. While a short list of survey species that would allow rapidly completed dives is preferable, it is worth reconsidering the species selected for monitoring in light of the list identified by the BESTStep analysis as appropriate for benthic monitoring. An expanded list would also better serve the stake holder interest that drove the development of the Taputeranga Marine Reserve. Although the purpose of marine reserves in New Zealand is to further scientific research, interest in Taputeranga was also fuelled by local interest in restoring WSC to former levels of abundance and diversity (Gardner, 2008). Given this interest, it would seem appropriate to adopt a monitoring list that reflects and tracks the status of local benthic diversity.

The patterns identified for ecological processes were somewhat surprising. The results of the PERMDISP analyses of environmental contributions to benthic

community structure were perplexing and I cannot offer an explanation for the significant contribution of water quality parameters with such a narrow range of values. The range of observed environmental values along WSC corresponded with values reported in earlier studies (Russell, 2004; Helson et al., 2007). This continuity of values suggests a stability to basic water quality processes despite WSC proximity to harbour turbidity. .

Variation in beta diversity on such a small scale was interesting but driven perhaps more by abundances and natural spatial variation than large shifts in community structure or environmental processes (Robinson & Tull, 2000; Raffaelli et al., 2003). Future studies might repeat these analyses with a different set of variables with more success at identifying strong environmental contributors. To my knowledge, this is the first study of fine-scale variation in alpha and beta diversity; current literature focuses on identifying patterns in and drivers of biodiversity gradients at large, latitudinal scales or across regions (Vierros et al., 2011; Wernberg et al., 2011; Villamor & Becero, 2012) and the processes driving those patterns do not operate at local scales (Crist et al, 2003).

The lack of readily available literature on stand-alone drop camera work led to the use of an oversampled design driven by inexperience and lack of technical support. This oversampling was confirmed by the analyses of differences in seabed areas generated by sampling intensities. The most efficient method would use a stratified random sampling design based on areas identified by unsupervised classification of a recently completed multibeam survey of the WSC nearshore zone. Results would then be used to run supervised classifications to develop continuous coverage raster layers of the benthos (Penrose et al., 2005). Drop-camera surveys are simply more efficient and require fewer drops when designed to sample georeferenced classes derived from backscatter data.

Despite the drop-camera oversampling, two large populations of *Macrocystis pyrifera* in Lyall and Princess Bays were not captured by either dived or remote survey. This omission might have been avoided by a stratified camera survey design that would have incorporated these visually obvious patches of giant kelp. Given this macro-algae's ecosystem engineer status (Dayton, 1985; Dayton et al., 1992; Graham et al., 2007), a future survey series targeting this species' the three WSC populations might be appropriate.

Various challenges must be addressed in terms of equipment, identification skills, and data processing when designing surveys. Baseline surveys provided the necessary identification skills but lack of taxonomic expertise can limit the usefulness of video content, and a lack of field guide or access to taxonomic expertise hindered both these survey's early efforts. Also, lack of choice in drop-

camera frame design can influence the development of one's methods. Towed sleds and video sledges with surface-supplied light source, still cameras, and compass are desirable equipment options and are standard gear in most of the benthic mapping literature consulted (Davies, 2001; Humborstad et al, 2004). A previous attempt to use a towed sled along the WSC had not been successful due to the rugosity of WSC seabed so was not allowed for consideration (Robert Williamson, pers. comm.). The streamlined design of the camera frame used in this study, if not as functional as a towed sled, did minimize entrapment and withstood being dragged over bedrock. Automated image analysis might be possible for well-illuminated taxa with well-defined, distinct colours (Kohler & Gill, 2006), but in areas with low light-levels due to high levels of suspended sediment and high settled sediment obscuring key features, manual analysis is necessary.

Both dived and remote surveys identified the same spatial distribution trends in the macro-algal canopy cover, with *Ecklonia radiata* the most cosmopolitan, as predicted from its status as a key species in New Zealand rocky-reef communities (Shears & Babcock, 2007), to the highly clustered *Macrocystis pyrifera*. The differences in spatial correlation suggest that canopy species differ in their recruitment and settlement abilities. The higher frequencies of understorey red algae at eastern sites was also identified by both survey tools, but only the dived survey detected the spatial characteristics of the understorey *Caulerpa* spp. The dived survey also provided a description of mobile macro-invertebrate abundance and distribution that had strong spatial components. The remote survey did not capture this group.

The influences of physical habitat characteristics and geomorphology on community structure have been identified for macro-algae, bivalves, poriferans and other invertebrates, and fish (Talman et al. 2004; Tuya et al., 2008; Wernberg & Connell, 2008; Tuya et al., 2009; Cardenas et al. 2012). The effect of habitat instability fragmentation, and loss on species richness, abundance, and composition have been documented in temperate rocky-reef systems and identified as a factor in phase shifts, lower abundances, and a decline in species richness (Dobson et al, 2006; Perkol-Finkel et al., 2010). The combination of WSC's exposed shore, the likelihood of increasing numbers of extreme storm events, and the relatively small proportion of shallow rocky-reef habitat identified by the drop camera survey could contribute to eventual shifts in macro-algal forest cover and associated taxa. Space cleared by storms is susceptible to colonization by smaller-bodied opportunistic algal and invertebrate species, with effects travelling up the trophic web as resource availability declined (Suchanek, 1994; Perkol-Finkel et al., 2010).

These two surveys have produced a more comprehensive picture of nearshore rocky-reef epibenthic abundance, distribution, diversity, size, tools for estimating standing stock and productivity for key species, and patterns of spatial variability by functional group and for available habitat. Neither survey type could have identified all these patterns at the two scales studied. Remote surveys provide overviews of physically dominant taxa across large areas; dived surveys provide the opportunity to manipulate, measure, and resolve taxonomy to a fine level across small, shallow areas. Both are constrained by technology and limited by exposed climates. Neither supplants the other and the choice of which survey method is most appropriate depends on a range of factors, including the research question(s), the type of information required and how it will be processed and analyzed, the scale of the area of interest, the type of equipment and human resources available, availability of technical expertise, and financial constraints.

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Appendix A (Chapter 3). Voucher images of unidentified sessile macro-invertebrates observed during two annual baseline surveys of Wellington south coast's shallow rocky-reef communities (2008-09). All samples were collected between 5-12m during summer. Identifications are tentative when included.

Bryozoa A: *Membranipora membranacea*
Princess Bay

Bryozoa B: Encrusting

Barret Reef

Bryozoa C: Catanecellid?
1st Wash

Bryozoa D: *Hippellozoon novazelandiae*?

Red Rocks

Bryozoa E
1st Wash

Bryozoa F, G
Breaker Bay

Appendix A (Chapter 3). Voucher images of unidentified sessile macro-invertebrates, page 2.

Bryozoa G: Catenicellid?

← Red Rocks →

Hydrozoa A: *Amphisbetia bispinosa*?

Hydrozoa B

Ascidiacea A: colonial

Ascidiacea B: colonial

The Sirens

Ascidiacea C: colonial

The Sirens

Appendix A (Chapter 3). Voucher images of unidentified sessile macro-invertebrates, page 3.

Ascidiacea D: colonial

Breaker Bay

Ascidiacea D, variations

Breaker Bay

Ascidiacea E: solitary, type 1

Breaker Bay

Ascidiacea E: solitary, type 2

Breaker Bay

Porifera A: Massive encrusting

Porifera B: Massive encrusting

Appendix A (Chapter 3). Voucher images of unidentified sessile macro-invertebrates, page 4.

Porifera C: Encrusting

The Sirens

Porifera E

Porifera D: Lobate, *Dysidea* sp.?

Porifera F: Massive vase, *Ancorina* sp.?

1st Wash

Appendix A (Chapter 3). Voucher images of unidentified sessile macro-invertebrates, page 5.

Porifera G: Papillate, *Sycella* sp.?

Breaker Bay

Porifera H: Massive globular, *Latrunculia* sp.?

The Sirens

Porifera I: Globulose, probably *Tethya* sp.

Porifera J:
Encrusting, *Hamigera tarangaensis*?

Red Rocks

Appendix A (Chapter 3). Voucher images of unidentified sessile macro-invertebrates, page 6.

Porifera K: Papillate, *Polymastia* sp. 1

Barret Reef

Porifera K: Papillate, *Polymastia* sp. 2

Princess Bay

Porifera L: Encrusting (*Tethya* sp., lower right)

Porifera M: Clathrate, globular, *Clathrina* sp.?

Porifera N: Repent, *Callyspongia* sp.?

Porifera O: Flask, *Leucosetta* sp.?

Appendix A (Chapter 3). Voucher images of unidentified sessile macro-invertebrates, final.

Porifera: *Tethya* sp. A Breaker Bay

Porifera: *Tethya* sp. B Red Rocks

Porifera: *Tethya* sp. C Red Rocks

Appendix B (Chapter 3). Voucher images of unidentified macro-algae observed during two annual baseline surveys of Wellington south coast's shallow rocky-reef communities (2008-09). All samples were collected between 5-12m during summer.

Rhodophyta: Turfing foliose

1st Wash

Rhodophyta: Turfing filamentous

Yung Pen

Rhodophyta: Understory foliose, Species A

Rhodophyta: Understory, foliose, Species B

↑ Red Rocks →

Appendix B (Chapter 3). Voucher images of unidentified macro-algae, page 2.

Rhodophyta: Understory foliose and blade, inset, Species C (*Gigartina* sp.?)

Rhodophyta: Understory filamentous, species A

Breaker Bay

Rhodophyta: Understory filamentous, species B

Red Rocks

Appendix B (Chapter 3). Voucher images of unidentified macro-algae, final. *Not shown*: Two additional understory Phaeophyta species observed once at one eastern site during the first annual survey.

Rhodophyta: Understory, foliose species C, and detail
(Chlorophyta *Chaetomorpha* sp. intertwined)

Phaeophyta: Understory foliose

Princess Bay

Phaeophyta: Turfing foliose (*Zonaria* sp.?)

2nd Wash

Phaeophyta? Understory filamentous, species A

Appendix C (chapter 3). Mean abundance of **macro-algae** (counts $m^{-2} \pm SE$, minimum (min) and maximum (max), subset of 11 species, from two annual baseline surveys of Wellington south coast's rocky-reef epibenthic communities (5-12m depth). Surveys were conducted during summer at nine sites in 2008 and ten sites in 2009 (Breaker Bay added). See Table 3.x for full taxonomic classification. Sites are listed from west (2nd Wash) to east (Barret Reef).

Survey year	Species	2nd Wash		1st Wash		Red Rocks		Yung Pen		The Sirens	
		Mean, SE	Min, Max	Mean, SE	Min, Max	Mean, SE	Min, Max	Mean, SE	Min, Max	Mean, SE	Min, Max
2008	<i>Carpophyllum flexuosum</i>	0.2 ± 0.2	(0, 2)	0.1 ± 0.1	(0, 1)	0.1 ± 0.5	(0, 5)	1.9 ± 0.8	(0, 7)	-	-
2008	<i>Carpophyllum maschalocarpum</i>	-	-	2.7 ± 0.9	(0, 7)	2.7 ± 0.9	(0, 7)	2.4 ± 1.1	(0, 11)	1.7 ± 1.1	(0, 11)
2008	<i>Cystophora scalaris</i>	-	-	-	-	-	-	0.3 ± 0.2	(0, 1)	0.8 ± 0.6	(0, 7)
2008	<i>Ecklonida radiata</i>	2.3 ± 0.7	(0, 6)	5.5 ± 1.0	(1, 9)	5.5 ± 1.0	(0, 1)	0.9 ± 0.2	(0, 2)	1.1 ± 0.5	(0, 5)
2008	<i>Glossophora kunthii</i>	-	-	-	-	0.1 ± 0.1	(0, 1)	1.2 ± 0.5	(0, 4)	0.2 ± 0.1	(0, 1)
2008	<i>Lessonia variegata</i>	1.3 ± 0.5	(0, 5)	0.1 ± 0.1	(0, 1)	2.5 ± 0.6	(0, 5)	1.5 ± 0.3	(0, 3)	0.2 ± 0.1	(0, 1)
2008	<i>Landsburgia quercifolia</i>	1.9 ± 0.6	(0, 5)	1.1 ± 0.4	(0, 4)	0.9 ± 0.5	(0, 4)	1.2 ± 0.7	(0, 6)	1.1 ± 0.5	(0, 4)
2008	<i>Macrocystis pyrifera</i>	-	-	-	-	-	-	-	-	-	-
2008	<i>Marginariella</i> spp.	6.7 ± 2.7	(0, 26)	0.2 ± 0.2	(0, 2)	-	-	0.1 ± 0.1	(0, 1)	-	-
2008	<i>Sargassum sinclairii</i>	-	-	0.1 ± 0.1	(0, 1)	-	-	0.5 ± 0.2	(0, 2)	1.2 ± 0.5	(0, 4)
2008	<i>Undaria pinnatifida</i>	-	-	-	-	-	-	0.6 ± 0.3	(0, 2)	-	-
2009	<i>Carpophyllum flexuosum</i>	0.2 ± 0.2	(0, 2)	2 ± 0.9	(0, 8)	0.6 ± 0.3	(0, 2)	2.7 ± 1.1	(0, 10)	0.2 ± 0.2	(0, 2)
2009	<i>Carpophyllum maschalocarpum</i>	-	-	-	-	4.4 ± 2.2	(0, 18)	5.6 ± 3.5	(0, 36)	1.6 ± 0.8	(0, 8)
2009	<i>Cystophora scalaris</i>	-	-	-	-	-	-	0.1 ± 0.1	(0, 1)	-	-
2009	<i>Ecklonida radiata</i>	6.3 ± 1.0	(0, 12)	5.1 ± 2.5	(0, 26)	1.7 ± 0.8	(0, 8)	2.1 ± 0.6	(0, 6)	1.8 ± 0.6	(0, 6)
2009	<i>Glossophora kunthii</i>	-	-	-	-	-	-	1 ± 0.5	(0, 5)	0.2 ± 0.2	(0, 2)
2009	<i>Lessonia variegata</i>	1.6 ± 0.4	(0, 2)	0.5 ± 0.2	(0, 2)	1.2 ± 0.3	(0, 3)	1.2 ± 0.4	(0, 4)	1.6 ± 0.3	(0, 3)
2009	<i>Landsburgia quercifolia</i>	1.2 ± 0.6	(0, 6)	0.7 ± 0.3	(0, 3)	0.5 ± 0.5	(0, 5)	2.1 ± 0.8	(0, 9)	2.2 ± 0.6	(1, 7)
2009	<i>Macrocystis pyrifera</i>	-	-	-	-	-	-	-	-	-	-
2009	<i>Marginariella</i> spp.	3.1 ± 1.5	(0, 14)	-	-	0.3 ± 0.2	(0, 2)	0.7 ± 0.4	(0, 4)	0.5 ± 0.3	(0, 3)
2009	<i>Sargassum sinclairii</i>	-	-	0.1 ± 0.1	(0, 1)	0.2 ± 0.1	(0, 1)	-	-	-	-
2009	<i>Undaria pinnatifida</i>	-	-	-	-	-	-	0.6 ± 0.3	(0, 3)	-	-

Appendix C (chapter 3). Mean abundances by counts of macro-algae, final.

Species	Princess Bay		Palmer Head		Flax Bay		Breaker Bay		Barret Reef		N
	Mean, SE	Min, Max	Mean, SE	Min, Max	Mean, SE	Min, Max	Mean, SE	Min, Max	Mean, SE	Min, Max	
<i>Carpophyllum flexuosum</i>	-	-	0.3 ± 0.2	(0, 2)	0.6 ± 0.6	(0, 6)	-	-	-	-	10
<i>Carpophyllum maschalocarpum</i>	1.1 ± 0.9	(0, 9)	1.6 ± 0.8	(0, 7)	2.6 ± 1.3	(0, 14)	-	-	1.00 ± 0.5	(0, 5)	10
<i>Cystophora scalaris</i>	-	-	-	-	0.1 ± 0.1	(0, 1)	-	-	-	-	10
<i>Ecklonida radiata</i>	0.5 ± 0.3	(0, 2)	4.2 ± 1.1	(0, 11)	0.7 ± 0.3	(0, 3)	-	-	8.3 ± 1.64	(1, 16)	10
<i>Glossophora kunthii</i>	0.1 ± 0.1	(0, 1)	0.2 ± 0.1	(0, 1)	0.1 ± 0.1	(0, 1)	-	-	-	-	10
<i>Lessonia variegata</i>	0.6 ± 0.2	(0, 1)	-	-	-	-	-	-	0.6 ± 0.3	(0, 3)	10
<i>Landsburgia quercifolia</i>	0.2 ± 0.1	(0, 1)	0.5 ± 0.2	(0, 2)	-	-	-	-	1.5 ± 0.5	(0, 5)	10
<i>Macrocystis pyrifera</i>	-	-	0.3 ± 0.1	(0, 2)	2.2 ± 0.7	(0, 7)	-	-	0.7 ± 0.6	(0, 6)	10
<i>Marginariella</i> spp.	-	-	-	-	-	-	-	-	0.8 ± 0.4	(0, 4)	10
<i>Sargassum sinclairii</i>	0.5 ± 0.3	(0, 3)	0.1 ± 0.1	(0, 1)	-	-	-	-	-	-	10
<i>Undaria pinnatifida</i>	0.1 ± 0.1	(0, 1)	-	-	0.5 ± 0.2	(0, 2)	-	-	-	-	10
<i>Carpophyllum flexuosum</i>	0.5 ± 0.4	(0, 3)	0.1 ± 0.1	(0, 1)	6.5 ± 2.5	(0, 26)	0.1 ± 0.1	(0, 1)	3.4 ± 1.4	(0, 12)	10
<i>Carpophyllum maschalocarpum</i>	1.9 ± 0.6	(0, 6)	-	-	3.9 ± 1.4	(0, 12)	8.5 ± 1.9	(4, 25)	0.6 ± 0.3	(0, 2)	10
<i>Cystophora scalaris</i>	-	-	0.1 ± 0.1	(0, 1)	-	-	-	-	-	-	10
<i>Ecklonida radiata</i>	0.9 ± 0.4	(0, 4)	5.4 ± 0.8	(2, 8)	2.3 ± 0.9	(0, 8)	5.8 ± 0.8	(2, 10)	9.4 ± 1.19	(3, 16)	10
<i>Glossophora kunthii</i>	1.5 ± 0.8	(0, 6)	1 ± 0.8	(0, 8)	0.4 ± 0.4	(0, 4)	2.1 ± 0.9	(0, 7)	-	-	10
<i>Lessonia variegata</i>	1.7 ± 0.5	(0, 4)	0.1 ± 0.1	(0, 1)	-	-	0.1 ± 0.1	(0, 1)	-	-	10
<i>Landsburgia quercifolia</i>	2.4 ± 0.9	(0, 9)	1.8 ± 0.6	(0, 6)	0.1 ± 0.1	(0, 1)	0.9 ± 0.5	(0, 4)	0.5 ± 0.2	(0, 2)	10
<i>Macrocystis pyrifera</i>	-	-	-	-	1.1 ± 0.4	(0, 4)	0.1 ± 0.1	(0, 1)	0.1 ± 0.1	(0, 1)	10
<i>Marginariella</i> spp.	0.8 ± 0.7	(0, 7)	-	-	-	-	-	-	1.1 ± 0.9	(0, 9)	10
<i>Sargassum sinclairii</i>	-	-	-	-	0.7 ± 0.3	(0, 3)	-	-	-	-	10
<i>Undaria pinnatifida</i>	-	-	-	-	0.2 ± 0.1	(0, 1)	-	-	-	-	10

Appendix D (chapter 3). Mean abundance of percent cover macro-algae ($m^{-2} \pm SE$, minimum (min) and maximum (max), all species in survey) surveyed for annual baseline surveys of Wellington south coast's rocky-reef epibenthic communities (5-12m depth). Surveys were conducted at nine sites in 2008 and ten sites in 2009. See Table 3.x for full taxonomic classification. Sites are listed here from west (2nd Wash) to east (Barret Reef). *Macro-algal species for which count data are presented in the "macro-algal abundances" table.

Survey year	Species	2 nd Wash		1 st Wash		Red Rocks		Yung Pen		The Sirens	
		Mean, SE	Min, Max	Mean, SE	Min, Max	Mean, SE	Min, Max	Mean, SE	Min, Max	Mean, SE	Min, Max
2008	<i>Asparagopsis armata</i>	-	-	-	-	-	-	-	-	-	-
2008	<i>Carpomitra costata</i>	0.1 ± 0.1	(0, 1)	1.3 ± 0.7	(0, 6)	0.1 ± 0.1	(0, 1)	0.6 ± 0.4	(0, 5)	0.1 ± 0.1	(0, 1)
2008	<i>Carpophyllum flexuosum</i> *	0.2 ± 0.2	(0, 2)	-	-	0.7 ± 0.5	(0, 5)	8 ± 3.2	(0, 30)	-	-
2008	<i>Carpophyllum maschalocarpum</i> *	-	-	11.4 ± 4.4	(0, 40)	3.2 ± 1.3	(0, 12)	4 ± 2.4	(0, 25)	3.4 ± 2.5	(0, 25)
2008	<i>Caulerpa geminata</i>	-	-	0.7 ± 0.5	(0, 6)	-	-	-	-	0.5 ± 0.5	(0, 0.5)
2008	<i>Caulerpa</i> spp. (brownii, flexilis)	7.7 ± 6.9	(0, 70)	24.1 ± 9.3	(0, 80)	5.3 ± 2.4	(0, 22)	15.3 ± 7.2	(0, 64)	26.6 ± 11.3	(0, 95)
2008	<i>Chaetomorpha intestinalis</i>	-	-	-	-	0.1 ± 0.1	(0, 1)	-	-	-	-
2008	<i>Champia laindii</i>	-	-	-	-	-	-	-	-	-	-
2008	<i>Cladhymenia oblongifolia</i>	-	-	-	-	-	-	2.5 ± 1.1	(0, 10)	-	-
2008	<i>Colpomenia sinuosa</i>	-	-	-	-	-	-	-	-	-	-
2008	Crustose coralline algae	65.6 ± 14.3	(2, 100)	85.7 ± 3.8	(60, 100)	96.5 ± 1.6	(85, 100)	89.6 ± 2.7	(70, 98)	38.2 ± 11.9	(0, 90)
2008	<i>Cystophora scalaris</i> *	-	-	0.1 ± 0.1	(0, 1)	-	-	0.4 ± 0.2	(0, 2)	3.1 ± 2.9	(0, 30)
2008	<i>Desmarestia ligulata</i>	-	-	-	-	-	-	-	-	-	-
2008	<i>Ecklonia radiata</i> *	7.7 ± 1.94	(0, 16)	39.6 ± 9.2	(10, 95)	0.5 ± 0.5	(0, 5)	4.4 ± 2.7	(0, 20)	4.6 ± 2.9	(0, 16)
2008	<i>Euptilota formisissima</i>	0.8 ± 0.4	(0, 4)	5.1 ± 1.4	(0, 12)	1.6 ± 0.6	(0, 6)	1.3 ± 0.6	(0, 5)	-	-
2008	Geniculate coralline algae	4.7 ± 1.3	(0, 12)	0.6 ± 0.6	(0, 6)	5.9 ± 1.4	(1, 15)	3.1 ± 0.8	(0, 7)	2 ± 1.5	(0, 16)
2008	<i>Gigartina circumcincta</i>	-	-	-	-	-	-	-	-	-	-
2008	<i>Gigartina decipiens</i>	-	-	-	-	-	-	-	-	-	-
2008	<i>Glossophora kunthii</i> *	-	-	-	-	0.1 ± 0.1	(0, 1)	1.3 ± 0.5	(0, 6)	0.2 ± 0.1	(0, 1)
2008	<i>Halopteris</i> sp.	1.4 ± 1.3	(0, 10)	3.9 ± 1.4	(0, 12)	0.3 ± 0.2	(0, 2)	7.6 ± 2.6	(0, 25)	4.9 ± 2.1	(0, 20)
2008	<i>Landsburgia quercifolia</i> *	4.1 ± 1.5	(0, 15)	2.1 ± 1.2	(0, 12)	2 ± 0.9	(0, 7)	2 ± 1.5	(0, 16)	1.2 ± 0.6	(0, 5)
2008	<i>Lessonia variegata</i> *	17.1 ± 7.9	(0, 80)	1 ± 1	(0, 100)	38.3 ± 9.7	(0, 100)	32.8 ± 8.4	(0, 70)	7 ± 4.7	(0, 40)
2008	<i>Macrocystis pyrifera</i> *	-	-	-	-	-	-	-	-	-	-
2008	<i>Marginariella</i> spp (<i>boryana</i> , <i>urvillia</i>)*	34.9 ± 14.3	(0, 100)	0.5 ± 0.5	(0, 5)	-	-	0.1 ± 0.1	(0, 1)	-	-
2008	Phaeophyta, unknown turfing A	-	-	-	-	-	-	-	-	-	-
2008	Phaeophyta, unknown turfing B	-	-	-	-	-	-	-	-	-	-
2008	Phaeophyta, unknown understory	-	-	-	-	-	-	-	-	-	-

Appendix D (chapter 3). Macro-algal percent cover², page 2.

Survey year	Species	Princess Bay		Palmer Head		Flax Bay		Breaker Bay		Barret Reef		N
		Mean, SE	Min, Max	Mean, SE	Min, Max	Mean, SE	Min, Max	Mean, SE	Min, Max	Mean, SE	Min, Max	
2008	<i>Asparagopsis armata</i>	-	-	14.1 ± 8.3	(0, 80)	0.9 ± 0.6	(0, 5)	-	-	-	-	10
2008	<i>Carpomitra costata</i>	0.1 ± 0.1	(0, 1)	-	-	0.1 ± 0.1	(0, 1)	-	-	-	-	10
2008	<i>Carpophyllum flexuosum</i> *	-	-	0.5 ± 0.4	(0, 4)	4 ± 4	(0, 40)	-	-	-	-	10
2008	<i>Carpophyllum maschalocarpum</i> *	2.7 ± 2.4	(0, 25)	5 ± 3.1	(0, 30)	12.4 ± 7.7	(0, 80)	-	-	3.1 ± 2.1	(0, 20)	10
2008	<i>Caulerpa geminata</i>	-	-	-	-	-	-	-	-	-	-	10
2008	<i>Caulerpa</i> spp. (brownii, flexilis)	38.7 ± 1.2	(0, 90)	-	-	-	-	-	-	-	-	10
2008	<i>Chaetomorpha intestinalis</i>	-	-	-	-	-	-	-	-	-	-	10
2008	<i>Champia laindii</i>	-	-	0.3 ± 0.2	(0, 1)	-	-	-	-	-	-	10
2008	<i>Cladhymenia oblongifolia</i>	-	-	0.8 ± 0.8	(0, 8)	4 ± 2.9	(0, 30)	-	-	3.3 ± 3.3	(0, 33)	10
2008	<i>Colpomenia sinuosa</i>	-	-	0.4 ± 0.1	(0, 1)	-	-	-	-	-	-	10
2008	Crustose coralline algae	47.6 ± 11.9	(0, 100)	61.6 ± 8.7	(20, 96)	3.2 ± 8.3	(0, 80)	-	-	35.2 ± 12.5	(1, 96)	10
2008	<i>Cystophora scalaris</i> *	-	-	-	-	0.1 ± 0.1	(0, 1)	-	-	-	-	10
2008	<i>Desmarestia ligulata</i>	-	-	-	-	-	-	-	-	-	-	10
2008	<i>Ecklonia radiata</i> *	4.3 ± 3.3	(0, 30)	24.3 ± 8.2	(0, 70)	5.2 ± 2.3	(0, 20)	-	-	58 ± 6.2	(25, 85)	10
2008	<i>Euptilota formisissima</i>	-	-	-	-	1.4 ± 1.3	(0, 10)	-	-	-	-	10
2008	Geniculate coralline algae	12.1 ± 3.2	(0, 30)	4.9 ± 1.2	(1, 11)	1.5 ± 0.6	(0, 6)	-	-	3.7 ± 1.5	(0, 9)	10
2008	<i>Gigartina circumcincta</i>	0.2 ± 0.2	(0, 2)	-	-	9.7 ± 5.8	(0, 60)	-	-	-	-	10
2008	<i>Gigartina decipiens</i>	-	-	-	-	0.8 ± 0.4	(0, 4)	-	-	-	-	10
2008	<i>Glossophora kunthii</i> *	0.1 ± 0.1	(0, 1)	1.4 ± 0.4	(0, 5)	0.1 ± 0.1	(0, 1)	-	-	-	-	10
2008	<i>Halopteris</i> sp.	4.3 ± 3.9	(0, 30)	0.7 ± 0.4	(0, 4)	0.4 ± 0.4	(0, 4)	-	-	-	-	10
2008	<i>Landsburgia quercifolia</i> *	0.5 ± 0.4	(0, 3)	0.4 ± 0.1	(0, 1)	-	-	-	-	2.5 ± 0.8	(0, 6)	10
2008	<i>Lessonia variegata</i> *	14 ± 7.7	(0, 70)	-	-	-	-	-	-	8.6 ± 4.6	(0, 35)	10
2008	<i>Macrocyctis pyrifera</i> *	-	-	0.5 ± 0.3	(0, 2)	5.3 ± 1.2	(0, 10)	-	-	1.4 ± 0.9	(0, 8)	10
2008	<i>Marginariella</i> spp (boryana, urvillia)*	-	-	-	-	-	-	-	-	2.4 ± 1.6	(0, 16)	10
2008	Phaeophyta, unknown turfing A	-	-	0.1 ± 0.1	(0, 1)	0.1 ± 0.1	(0, 1)	-	-	-	-	10
2008	Phaeophyta, unknown turfing B	-	-	1.2 ± 0.8	(0, 8)	-	-	-	-	-	-	10
2008	Phaeophyta, unknown understory	-	-	0.2 ± 0.1	(0, 1)	-	-	-	-	-	-	10

Appendix D (chapter 3). Macro-algal percent cover, page 3.

Survey year	Species	2 nd Wash		1 st Wash		Red Rocks		Yung Pen		The Sirens	
		Mean, SE	Min, Max	Mean, SE	Min, Max	Mean, SE	Min, Max	Mean, SE	Min, Max	Mean, SE	Min, Max
2008	<i>Plocamium microcladioides</i>	-	-	-	-	2.3 ± 2.1	(0, 22)	-	-	-	-
2008	<i>Pterocladia lucida</i>	-	-	-	-	0.4 ± 0.2	(0, 2)	0.2 ± 0.1	(0, 1)	0.1 ± 0.1	(0, 1)
2008	Rhodophyta, unidentified turfing A	-	-	-	-	-	-	-	-	-	-
2008	Rhodophyta, unidentified turfing B	1.4 ± 0.8	(0, 8)	2.2 ± 1.2	(0, 12)	1.8 ± 0.8	(0, 9)	1.6 ± 0.6	(0, 6)	-	-
2008	Rhodophyta, unidentified understory A	-	-	-	-	0.1 ± 0.1	(0, 1)	-	-	-	-
2008	Rhodophyta, unidentified understory B	0.2 ± 0.2	(0, 2)	-	-	0.5 ± 0.5	(0, 0.5)	0.2 ± 0.2	(0, 2)	0.3 ± 0.1	(0, 1)
2008	Rhodophyta, unidentified understory C	-	-	-	-	0.1 ± 0.1	(0, 1)	-	-	0.2 ± 0.2	(0, 2)
2008	Rhodophyta, unidentified understory D	-	-	-	-	-	-	1.3 ± 0.3	(0, 4)	0.6 ± 0.4	(0, 4)
2008	Rhodophyta, unidentified understory E	0.4 ± 0.2	(0, 1)	0.1 ± 0.1	(0, 1)	0.5 ± 0.5	(0, 5)	-	-	-	-
2008	Rhodophyta, unidentified understory F	-	-	-	-	-	-	-	-	-	-
2008	<i>Rhodomenia dichotoma</i>	-	-	-	-	-	-	-	-	-	-
2008	<i>Sargassum sinclairii</i> *	-	-	0.1 ± 0.1	(0, 1)	-	-	0.3 ± 0.2	(0, 2)	1.9 ± 0.7	(0, 5)
2008	<i>Ulva lactuca</i>	-	-	-	-	-	-	0.8 ± 0.8	(0, 8)	2.5 ± 2.5	(0, 25)
2008	<i>Undaria pinnatifida</i> *	-	-	-	-	-	-	1.8 ± 0.9	(0, 8)	-	-
2008	<i>Xiphophora gladiata</i>	-	-	-	-	0.1 ± 0.1	(0, 1)	-	-	0.5 ± 0.5	(0, 0.5)
2008	<i>Zonaria</i> , turfing	-	-	-	-	-	-	-	-	0.2 ± 0.1	(0, 1)
2008	<i>Zonaria turneriana</i>	4.1 ± 2.5	(0, 20)	6.2 ± 2.2	(0, 20)	1.6 ± 0.5	(0, 4)	2.3 ± 1.2	(0, 10)	2.3 ± 2.1	(0, 22)
2009	<i>Asparagopsis armata</i>	-	-	-	-	-	-	-	-	-	-
2009	<i>Carpomitra costata</i>	0.2 ± 0.2	(0, 1)	0.6 ± 0.4	(0, 4)	0.1 ± 0.1	(0, 1)	-	-	0.7 ± 0.3	(0, 3)
2009	<i>Carpophyllum flexuosum</i>	0.2 ± 0.2	(0, 2)	4.1 ± 1.4	(0, 12)	5.3 ± 3.4	(0, 35)	9.3 ± 7.3	(0, 72)	0.1 ± 0.1	(0, 1)
2009	<i>Carpophyllum maschalocarpum</i>	-	-	-	-	2.9 ± 1.5	(0, 12)	6.8 ± 5.5	(0, 52)	1.4 ± 0.7	(0, 7)
2009	<i>Caulerpa geminata</i>	-	-	2.2 ± 2.2	(0, 22)	-	-	-	-	1.9 ± 1.9	(0, 19)
2009	<i>Caulerpa</i> spp. (brownii, flexilis)	2.1 ± 1.8	(0, 18)	36.7 ± 12.3	(0, 96)	24.8 ± 1.4	(0, 85)	3.25 ± 2.9	(0, 30)	3.85 ± 1.8	(0, 19)
2009	<i>Chaetomorpha intestinalis</i>	-	-	-	-	-	-	-	-	-	-
2009	<i>Champia laindii</i>	-	-	-	-	-	-	-	-	-	-
2009	<i>Cladhymenia oblongifolia</i>	-	-	-	-	0.1 ± 0.1	(0, 1)	-	-	0.2 ± 0.2	(0, 2)
2009	<i>Colpomenia sinuosa</i>	-	-	-	-	-	-	-	-	-	-
2009	Crustose coralline algae	95.5 ± 1.9	(83, 100)	79.1 ± 7.2	(35, 100)	8.7 ± 1.3	(0, 100)	86.5 ± 7.8	(20, 100)	91.8 ± 2.2	(80, 100)

Appendix D (chapter 3). Macro-algal percent cover, page 4.

Survey year	Species	Princess Bay		Palmer Head		Flax Bay		Breaker Bay		Barret Reef		N
		Mean, SE	Min, Max	Mean, SE	Min, Max	Mean, SE	Min, Max	Mean, SE	Min, Max	Mean, SE	Min, Max	
2008	<i>Plocamium microcladioides</i>	-	-	-	-	-	-	-	-	-	-	10
2008	<i>Pterocladia lucida</i>	-	-	-	-	-	-	-	-	-	-	10
2008	Rhodophyta, unidentified turfing A	0.2 ± 0.2	(0, 2)	1.8 ± 1.1	(0, 12)	0.3 ± 0.3	(0, 3)	-	-	-	-	10
2008	Rhodophyta, unidentified turfing B	0.6 ± 0.3	(0, 3)	2.5 ± 1.7	(0, 18)	-	-	-	-	6.1 ± 2.1	(0, 22)	10
2008	Rhodophyta, unidentified understory A	-	-	-	-	-	-	-	-	-	-	10
2008	Rhodophyta, unidentified understory B	0.3 ± 0.2	(0, 2)	0.1 ± 0.1	(0, 1)	0.2 ± 0.1	(0, 1)	-	-	-	-	10
2008	Rhodophyta, unidentified understory C	0.9 ± 0.4	(0, 5)	0.4 ± 0.4	(0, 4)	0.1 ± 0.1	(0, 1)	-	-	-	-	10
2008	Rhodophyta, unidentified understory D	0.4 ± 0.4	(0, 4)	7.6 ± 2.7	(0, 25)	2 ± 1.1	(0, 9)	-	-	-	-	10
2008	Rhodophyta, unidentified understory E	-	-	0.4 ± 0.2	(0, 2)	-	-	-	-	-	-	10
2008	Rhodophyta, unidentified understory F	-	-	0.2 ± 0.2	(0, 2)	-	-	-	-	-	-	10
2008	<i>Rhodymenia dichotoma</i>	-	-	-	-	-	-	-	-	-	-	10
2008	<i>Sargassum sinclairii</i> *	0.3 ± 0.1	(0, 1)	0.4 ± 0.4	(0, 4)	-	-	-	-	-	-	10
2008	<i>Ulva lactuca</i>	-	-	0.2 ± 0.1	(0, 1)	3.1 ± 1.5	(0, 10)	-	-	-	-	10
2008	<i>Undaria pinnatifida</i> *	0.1 ± 0.1	(0, 1)	-	-	5.6 ± 2.4	(0, 20)	-	-	-	-	10
2008	<i>Xiphophora gladiata</i>	-	-	-	-	-	-	-	-	-	-	10
2008	<i>Zonaria</i> , turfing	0.3 ± 0.2	(0, 1)	2.8 ± 1.6	(0, 16)	-	-	-	-	-	-	10
2008	<i>Zonaria turneriana</i>	0.7 ± 0.3	(0, 4)	11.6 ± 2.9	(0, 27)	0.1 ± 0.1	(0, 1)	-	-	0.2 ± 0.1	(0, 1)	10
2009	<i>Asparagopsis armata</i>	-	-	0.8 ± 0.5	(0, 5)	0.5 ± 0.3	(0, 2)	-	-	-	-	10
2009	<i>Carpomitra costata</i>	-	-	0.5 ± 0.2	(0, 2)	-	-	-	-	0.2 ± 0.1	(0, 1)	10
2009	<i>Carpophyllum flexuosum</i>	0.11 ± 0.9	(0, 1)	0.1 ± 0.1	(0, 1)	12.4 ± 3.7	(0, 32)	0.1 ± 0.1	(0, 1)	8.9 ± 3.8	(0, 33)	10
2009	<i>Carpophyllum maschalocarpum</i>	0.91 ± 0.5	(0, 5)	0.15 ± 0.2	(0, 1)	6.4 ± 2.9	(0, 26)	13.2 ± 3.9	(2, 38)	1.31 ± 0.8	(0, 7)	10
2009	<i>Caulerpa geminata</i>	0.5 ± 0.5	(0, 0.5)	-	-	-	-	-	-	-	-	10
2009	<i>Caulerpa</i> spp. (brownii, flexilis)	1.55 ± 1.7	(0, 11)	-	-	-	-	-	-	-	-	10
2009	<i>Chaetomorpha intestinalis</i>	-	-	-	-	-	-	-	-	-	-	10
2009	<i>Champia laindii</i>	-	-	-	-	-	-	-	-	-	-	10
2009	<i>Cladhymenia oblongifolia</i>	-	-	-	-	-	-	0.6 ± 0.4	(0, 4)	-	-	10
2009	<i>Colpomenia sinuosa</i>	-	-	-	-	-	-	-	-	-	-	10
2009	Crustose coralline algae	83.5 ± 4.4	(55, 100)	72.7 ± 12.6	(0, 100)	6.8 ± 11.6	(0, 92)	91.5 ± 4.1	(62, 100)	9.8 ± 1.7	(78, 97)	10

Appendix D (chapter 3). Macro-algal percent cover, page 5.

Survey year	Species	2 nd Wash		1 st Wash		Red Rocks		Yung Pen		The Sirens	
		Mean, SE	Min, Max	Mean, SE	Min, Max	Mean, SE	Min, Max	Mean, SE	Min, Max	Mean, SE	Min, Max
2009	<i>Cystophora scalaris</i>	-	-	-	-	-	-	0.5 ± 0.5	(0, 0.5)	-	-
2009	<i>Desmarestia ligulata</i>	0.5 ± 0.5	(0, 0.5)	0.1 ± 0.1	(0, 1)	-	-	1.6 ± 0.9	(0, 9)	1.1 ± 0.5	(0, 5)
2009	<i>Ecklonia radiata</i>	23.3 ± 4.3	(0, 42)	14.2 ± 8.2	(0, 85)	5.6 ± 1.9	(0, 18)	6.2 ± 1.7	(0, 16)	5.3 ± 1.9	(0, 16)
2009	<i>Euptilota formisissima</i>	4.2 ± 1.2	(0, 11)	1.6 ± 0.6	(0, 6)	2.1 ± 1.4	(0, 10)	1.3 ± 0.6	(0, 6)	1.3 ± 0.5	(0, 5)
2009	Geniculate coralline algae	2.7 ± 0.6	(0, 6)	0.1 ± 0.1	(0, 1)	2.3 ± 0.7	(0, 6)	4.31 ± 1.3	(0, 12)	5.5 ± 2.2	(0, 18)
2009	<i>Gigartina circumcincta</i>	-	-	-	-	-	-	-	-	0.4 ± 0.4	(0, 4)
2009	<i>Gigartina decipiens</i>	-	-	-	-	-	-	0.1 ± 0.1	(0, 1)	-	-
2009	<i>Glossophora kunthii</i>	0.1 ± 0.1	(0, 1)	-	-	-	-	1.5 ± 0.5	(0, 5)	0.1 ± 0.1	(0, 0.1)
2009	<i>Halopteris</i> sp.	-	-	2.3 ± 1.6	(0, 16)	0.2 ± 0.1	(0, 1)	3.1 ± 0.9	(0, 8)	1.7 ± 0.7	(0, 6)
2009	<i>Landsburgia quercifolia</i>	3.7 ± 1.9	(0, 20)	0.6 ± 0.3	(0, 2)	1.31 ± 1.1	(0, 12)	5.2 ± 2.2	(0, 18)	2.7 ± 0.8	(0.1, 7)
2009	<i>Lessonia variegata</i>	42.5 ± 1.3	(0, 100)	15 ± 8.4	(0, 80)	23.4 ± 9.7	(0, 90)	16.7 ± 4.9	(0, 52)	34.3 ± 1.4	(0, 90)
2009	<i>Macrocyctis pyrifera</i>	-	-	-	-	-	-	-	-	-	-
2009	<i>Marginariella</i> spp (<i>boryana</i> , <i>urvillia</i>)	8.1 ± 4.6	(0, 45)	-	-	0.65 ± 0.5	(0, 5)	1.1 ± 0.7	(0, 8)	0.75 ± 0.5	(0, 5)
2009	Phaeophyta, unknown turfing A	-	-	-	-	-	-	-	-	-	-
2009	Phaeophyta, unknown turfing B	-	-	-	-	-	-	-	-	-	-
2009	Phaeophyta, unknown understory	-	-	-	-	-	-	-	-	-	-
2009	<i>Plocamium microcladioides</i>	0.1 ± 0.1	(0, 1)	-	-	0.1 ± 0.1	(0, 1)	0.5 ± 0.5	(0, 0.5)	-	-
2009	<i>Pterocladia lucida</i>	1.8 ± 0.9	(0, 9)	0.2 ± 0.1	(0, 1)	0.2 ± 0.2	(0, 2)	0.2 ± 0.2	(0, 2)	0.7 ± 0.5	(0, 5)
2009	Rhodophyta, unidentified turfing A	-	-	-	-	-	-	-	-	-	-
2009	Rhodophyta, unidentified turfing B	0.8 ± 0.2	(0, 1.5)	-	-	-	-	0.4 ± 0.2	(0, 2)	0.25 ± 0.1	(0, 1)
2009	Rhodophyta, unidentified understory A	-	-	-	-	-	-	-	-	0.4 ± 0.3	(0, 3)
2009	Rhodophyta, unidentified understory B	-	-	-	-	-	-	-	-	-	-
2009	Rhodophyta, unidentified understory C	0.2 ± 0.1	(0, 1)	0.1 ± 0.1	(0, 1)	0.3 ± 0.3	(0, 3)	0.3 ± 0.3	(0, 3)	0.1 ± 0.1	(0, 1)
2009	Rhodophyta, unidentified understory D	-	-	-	-	-	-	0.5 ± 0.5	(0, 0.5)	0.7 ± 0.5	(0, 5)
2009	Rhodophyta, unidentified understory E	0.3 ± 0.2	(0, 2)	-	-	0.3 ± 0.2	(0, 2)	0.1 ± 0.6	(0, 0.5)	0.1 ± 0.1	(0, 0.1)
2009	Rhodophyta, unidentified understory F	-	-	-	-	-	-	-	-	-	-
2009	<i>Rhodymenia dichotoma</i>	0.1 ± 0.1	(0, 1)	-	-	-	-	0.1 ± 0.1	(0, 0.1)	-	-
2009	<i>Sargassum sinclairii</i>	-	-	-	-	0.5 ± 0.5	(0, 0.5)	-	-	-	-
2009	<i>Ulva lactuca</i>	-	-	0.1 ± 0.1	(0, 1)	0.1 ± 0.1	(0, 0.1)	1.65 ± 1.4	(0, 15)	-	-
2009	<i>Undaria pinnatifida</i>	-	-	-	-	-	-	2.5 ± 1.8	(0, 19)	-	-
2009	<i>Xiphophora gladiata</i>	-	-	-	-	-	-	-	-	-	-
2009	<i>Zonaria</i> , turfing	0.3 ± 0.2	(0, 1)	0.6 ± 0.6	(0, 6)	0.3 ± 0.3	(0, 3)	0.2 ± 0.1	(0, 1)	-	-
2009	<i>Zonaria turneriana</i>	0.3 ± 0.2	(0, 1.5)	0.6 ± 0.3	(0, 3)	1.8 ± 0.6	(0, 6)	1.85 ± 0.6	(0, 6)	2.4 ± 0.7	(0, 7)

Appendix D (chapter 3). Macro-algal percent cover, final.

Survey year	Species	Princess Bay		Palmer Head		Flax Bay		Breaker Bay		Barret Reef		N
		Mean, SE	Min, Max	Mean, SE	Min, Max	Mean, SE	Min, Max	Mean, SE	Min, Max	Mean, SE	Min, Max	
2009	<i>Cystophora scalaris</i>	-	-	-	-	-	-	-	-	-	-	10
2009	<i>Desmarestia ligulata</i>	1.6 ± 0.9	(0, 9)	3 ± 1.2	(0, 9)	0.6 ± 0.6	(0, 6)	2.25 ± 1.1	(0, 9)	0.8 ± 0.6	(0, 6)	10
2009	<i>Ecklonia radiata</i>	4.1 ± 1.9	(0, 20)	29.7 ± 8.5	(5, 82)	21.6 ± 1.7	(0, 90)	44.9 ± 9.3	(9, 88)	55.3 ± 11.4	(7, 98)	10
2009	<i>Euptilota formisissima</i>	-	-	5.6 ± 1.1	(2, 12)	0.2 ± 0.2	(0, 2)	-	-	0.2 ± 0.1	(0, 1)	10
2009	Geniculate coralline algae	11.2 ± 1.9	(4, 23)	4.5 ± 1.2	(1, 13)	2.5 ± 0.6	(0, 5)	3.8 ± 0.8	(1, 8)	2.4 ± 0.9	(0, 8)	10
2009	<i>Gigartina circumcincta</i>	-	-	-	-	-	-	-	-	-	-	10
2009	<i>Gigartina decipiens</i>	-	-	-	-	-	-	-	-	-	-	10
2009	<i>Glossophora kunthii</i>	4.3 ± 2.5	(0, 22)	1.4 ± 0.9	(0, 9)	0.4 ± 0.4	(0, 4)	1.9 ± 0.9	(0, 9)	-	-	10
2009	<i>Halopteris</i> sp.	0.8 ± 0.3	(0, 4)	0.6 ± 0.3	(0, 3)	-	-	-	-	0.3 ± 0.2	(0, 2)	10
2009	<i>Landsburgia quercifolia</i>	1.4 ± 0.6	(0, 5)	2.8 ± 1.6	(0, 8)	0.1 ± 0.1	(0, 1)	1.4 ± 0.8	(0, 7)	0.4 ± 0.3	(0, 2)	10
2009	<i>Lessonia variegata</i>	25.9 ± 9.4	(0, 92)	0.5 ± 0.5	(0, 5)	-	-	0.7 ± 0.7	(0, 7)	-	-	10
2009	<i>Macrocystis pyrifera</i>	-	-	-	-	5.8 ± 3.9	(0, 40)	7 ± 7	(0, 70)	0.1 ± 0.1	(0, 1)	10
2009	<i>Marginariella</i> spp (<i>boryana</i> , <i>urvillia</i>)	1.9 ± 1.7	(0, 18)	-	-	0.5 ± 0.5	(0, 0.5)	-	-	1.8 ± 1.6	(0, 17)	10
2009	Phaeophyta, unknown turfing A	-	-	-	-	-	-	-	-	-	-	10
2009	Phaeophyta, unknown turfing B	-	-	-	-	-	-	-	-	-	-	10
2009	Phaeophyta, unknown understory	-	-	-	-	-	-	-	-	-	-	10
2009	<i>Plocamium microcladioides</i>	0.1 ± 0.1	(0, 1)	0.2 ± 0.7	(0, 0.5)	-	-	-	-	0.5 ± 0.5	(0, 0.5)	10
2009	<i>Pterocladia lucida</i>	1 ± 0.8	(0, 9)	0.1 ± 0.1	(0, 1)	-	-	-	-	-	-	10
2009	Rhodophyta, unidentified turfing A	0.2 ± 0.2	(0, 2)	-	-	-	-	0.2 ± 0.2	(0, 2)	0.5 ± 0.5	(0, 0.5)	10
2009	Rhodophyta, unidentified turfing B	-	-	4.3 ± 2.7	(0, 28)	0.2 ± 0.1	(0, 1)	-	-	0.8 ± 0.2	(0, 2)	10
2009	Rhodophyta, unidentified understory A	-	-	0.1 ± 0.1	(0, 1)	0.15 ± 0.15	(0, 1.5)	-	-	-	-	10
2009	Rhodophyta, unidentified understory B	-	-	0.5 ± 0.5	(0, 0.5)	-	-	-	-	-	-	10
2009	Rhodophyta, unidentified understory C	0.3 ± 0.2	(0, 2)	0.5 ± 0.5	(0, 0.5)	1.9 ± 1.1	(0, 11)	-	-	-	-	10
2009	Rhodophyta, unidentified understory D	-	-	-	-	-	-	-	-	-	-	10
2009	Rhodophyta, unidentified understory E	-	-	-	-	0.1 ± 0.1	(0, 1)	-	-	-	-	10
2009	Rhodophyta, unidentified understory F	-	-	-	-	-	-	-	-	-	-	10
2009	<i>Rhodomenia dichotoma</i>	-	-	0.2 ± 0.1	(0, 1)	-	-	-	-	-	-	10
2009	<i>Sargassum sinclairii</i>	-	-	-	-	0.65 ± 0.3	(0, 2)	0.1 ± 0.1	(0, 1)	-	-	10
2009	<i>Ulva lactuca</i>	-	-	1.5 ± 0.8	(0, 7)	0.5 ± 0.5	(0, 0.5)	-	-	-	-	10
2009	<i>Undaria pinnatifida</i>	-	-	-	-	0.5 ± 0.3	(0, 3)	0.2 ± 0.2	(0, 2)	-	-	10
2009	<i>Xiphophora gladiata</i>	-	-	-	-	-	-	-	-	-	-	10
2009	<i>Zonaria</i> , turfing	-	-	0.1 ± 0.1	(0, 1)	0.1 ± 0.1	(0, 1)	1 ± 0.5	(0, 5)	-	-	10
2009	<i>Zonaria turneriana</i>	4.5 ± 2.1	(0, 18)	1.8 ± 0.6	(0, 6)	1.9 ± 1.2	(0, 12)	3.9 ± 1.8	(0, 17)	3.9 ± 1.6	(0, 15)	10

Appendix E (chapter 3). Mean abundances (counts $m^{-2} \pm SE$, minimum (min) and maximum (max)) of **mobile macro-invertebrates** surveyed for two annual baseline surveys of Wellington south coast's rocky-reef epibenthic communities (5-12m depth). Surveys were conducted by day during summer using 4 belt transects/site at nine sites in 2008 and ten sites in 2009 (Breaker Bay added). See Table 3.x for full taxonomic classification. Sites are presented here from west (2nd Wash) to east (Barret Reef).

Survey year	Species	2nd Wash		1st Wash		Red Rocks		Yung Pen		The Sirens	
		Mean, SE	Min, Max	Mean, SE	Min, Max	Mean, SE	Min, Max	Mean, SE	Min, Max	Mean, SE	Min, Max
2008	<i>Aphelodoris luctosa</i>	-	-	-	-	0.5 $\pm$ 0.5	(0, 2)	0.3 $\pm$ 0.3	(0, 1)	0.3 $\pm$ 0.3	(0, 1)
2008	<i>Archidoris wellingtoniensis</i>	-	-	-	-	-	-	-	-	-	-
2008	<i>Argobuccinum pustulosum tumidum</i>	-	-	-	-	-	-	-	-	-	-
2008	<i>Astrostele scabrus</i>	1 $\pm$ 0.7	(0, 3)	0.5 $\pm$ 0.3	(0, 1)	0.8 $\pm$ 0.5	(0, 2)	1.5 $\pm$ 0.9	(0, 4)	1.0 $\pm$ 0.7	(0, 3)
2008	<i>Buccilinum linea</i>	-	-	-	-	0.5 $\pm$ 0.5	(0, 2)	0 $\pm$ 0	(0, 0)	0.5 $\pm$ 0.3	(0, 1)
2008	<i>Cabestana spengleri</i>	-	-	-	-	-	-	-	-	-	-
2008	<i>Cantharidus purpureus</i>	-	-	-	-	-	-	-	-	1.0 $\pm$ 0.4	(0, 2)
2008	<i>Cantharidus opalus</i>	-	-	-	-	-	-	-	-	-	-
2008	<i>Chromodoris aureomarginata</i>	-	-	-	-	-	-	-	-	-	-
2008	<i>Cominella virgata</i>	-	-	-	-	-	-	0.3 $\pm$ 0.3	(0, 1)	-	-
2008	<i>Cookia sulcata</i>	0.8 $\pm$ 0.8	(0, 3)	-	-	1.8 $\pm$ 0.5	(1, 3)	1.5 $\pm$ 0.5	(0, 2)	0.3 $\pm$ 0.3	(0, 1)
2008	<i>Coscinasterias muricata</i>	-	-	-	-	-	-	-	-	-	-
2008	<i>Diplondontias dilatatus</i>	2.5 $\pm$ 1.0	(0, 5)	1.3 $\pm$ 0.5	(0, 2)	0.5 $\pm$ 0.3	(0, 1)	1.3 $\pm$ 0.8	(0, 3)	-	-
2008	<i>Evechinus chloroticus</i>	2.3 $\pm$ 0.8	(1, 4)	3.8 $\pm$ 1.2	(2, 7)	3.3 $\pm$ 1.1	(1, 6)	0.5 $\pm$ 0.3	(0, 1)	1.0 $\pm$ 0.4	(0, 2)
2008	<i>Haliotis australis</i>	0.5 $\pm$ 0.3	(0, 1)	1.0 $\pm$ 0.7	(0, 3)	1.8 $\pm$ 1.8	(0, 7)	3 $\pm$ 1.1	(1, 6)	2.5 $\pm$ 0.6	(1, 4)
2008	<i>Haliotis iris</i>	0.3 $\pm$ 0.3	(0, 1)	-	-	0.5 $\pm$ 0.3	(0, 1)	6 $\pm$ 1.7	(3, 9)	1.5 $\pm$ 0.9	(0, 4)
2008	<i>Haustrum haustorum</i>	-	-	-	-	1.0 $\pm$ 0.6	(0, 2)	0.5 $\pm$ 0.5	(0, 2)	0.5 $\pm$ 0.5	(0, 2)
2008	<i>Ischnochiton maorianus</i>	-	-	-	-	-	-	-	-	-	-
2008	<i>Jasus edwardsii</i>	0.5 $\pm$ 0.3	(0, 1)	0.8 $\pm$ 0.8	(0, 3)	3.3 $\pm$ 3.3	(0, 13)	0.5 $\pm$ 0.5	(0, 2)	3.3 $\pm$ 3.3	(0, 13)
2008	<i>Maoricolpus roseus</i>	-	-	-	-	-	-	0 $\pm$ 0	(0, 0)	-	-
2008	<i>Modelia granulosa</i>	-	-	-	-	-	-	-	-	-	-
2008	<i>Octopus vulgaris</i>	-	-	-	-	-	-	-	-	-	-
2008	<i>Patiriella regularis</i>	25.3 $\pm$ 10.5	(1, 52)	0.8 $\pm$ 0.3	(0, 1)	32.8 $\pm$ 12.7	(4, 56)	26.5 $\pm$ 8.9	(6, 46)	5.3 $\pm$ 3.4	(0, 15)

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Species	Princess Bay		Palmer Head		Flax Bay		Breaker Bay		Barret Reef	
	Mean, SE	Min, Max	Mean, SE	Min, Max	Mean, SE	Min, Max	Mean, SE	Min, Max	Mean, SE	Min, Max
<i>Aphelodoris luctosa</i>	0.5 ± 0.3	(0, 1)	-	-	-	-			0.3 ± 0.3	(0, 1)
<i>Archidoris wellingtoniensis</i>	-	-	-	-	-	-			-	-
<i>Argobuccinum pustulosum tumidum</i>	-	-	-	-	1.0 ± 0.7	(0, 3)			-	-
<i>Astrostole scabrus</i>	1.5 ± 0.6	(0, 3)	1.0 ± 0.6	(0, 2)	0.3 ± 0.3	(0, 1)			0.8 ± 0.5	(0, 2)
<i>Buccilinum linea</i>	0.3 ± 0.3	(0, 1)	-	-	-	-			0.5 ± 0.5	(0, 2)
<i>Cabestana spengleri</i>	-	-	-	-	-	-			0.5 ± 0.3	(0, 1)
<i>Cantharidus purpureus</i>	0.3 ± 0.3	(0, 1)	-	-	-	-			-	-
<i>Cantharidus opalus</i>	1 ± 1	(0, 4)	-	-	-	-			0.5 ± 0.3	(0, 1)
<i>Chromodoris aureomarginata</i>	0.8 ± 0.8	(0, 3)	-	-	-	-			-	-
<i>Cominella virgata</i>	0 ± 0	(0, 0)	0.3 ± 0.3	(0, 1)	1 ± 0.7	(0, 3)			-	-
<i>Cookia sulcata</i>	0.75 ± 0.75	(0, 3)	0.3 ± 0.3	(0, 1)	1.3 ± 0.5	(0, 2)			-	-
<i>Coscinasterias muricata</i>	0.3 ± 0.3	(0, 1)	-	-	-	-			-	-
<i>Diplondontias dilatatus</i>	0.3 ± 0.3	(0, 1)	-	-	-	-			0.5 ± 0.5	(0, 2)
<i>Evechinus chloroticus</i>	1.5 ± 0.6	(0, 3)	15.5 ± 8.6	(1, 37)	0.3 ± 0.3	(0, 1)			12.3 ± 3.1	(6, 18)
<i>Haliotis australis</i>	11.0 ± 3.4	(4, 19)	1.0 ± 0.6	(0, 2)	1.5 ± 0.9	(0, 4)			0.5 ± 0.3	(0, 1)
<i>Haliotis iris</i>	11.5 ± 1.6	(8, 15)	2.0 ± 1.7	(0, 7)	5.5 ± 3.4	(0, 15)			6.5 ± 3.2	(0, 15)
<i>Haustrum haustorum</i>	0.3 ± 0.3	(0, 1)	0.8 ± 0.5	(0, 2)	1.0 ± 0.6	(0, 2)			-	-
<i>Ischnochiton maorianus</i>	-	-	-	-	0.5 ± 0.5	(0, 2)			-	-
<i>Jasus edwardsii</i>	2.3 ± 1.0	(0, 5)	0.8 ± 0.8	(0, 3)	-	-			0.5 ± 0.3	(0, 1)
<i>Maoricolpus roseus</i>	-	-	-	-	0.3 ± 0.3	(0, 1)			0.3 ± 0.3	(0, 1)
<i>Modelia granulosa</i>	-	-	-	-	-	-			-	-
<i>Octopus vulgaris</i>	-	-	-	-	-	-			-	-
<i>Patiriella regularis</i>	74.8 ± 20.1	(39, 128)	65.8 ± 17.1	(18, 97)	58.3 ± 9.4	(33, 73)			4.8 ± 3.3	(0, 14)

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Survey year	Species	2nd Wash		1st Wash		Red Rocks		Yung Pen		The Sirens	
		Mean, SE	Min, Max	Mean, SE	Min, Max	Mean, SE	Min, Max	Mean, SE	Min, Max	Mean, SE	Min, Max
2008	<i>Pectinura maculata</i>	-	-	-	-	1.0 ± 0.6	(0, 2)	0.3 ± 0.3	(0, 1)	-	-
2008	<i>Penion sulcatus</i>	-	-	-	-	-	-	0.3 ± 0.3	(0, 1)	-	-
2008	<i>Pentagonaster pulchellus</i>	1 ± 0.4	(0, 2)	0.3 ± 0.3	(0, 1)	0.75 ± 0.5	(0, 2)	2.8 ± 1.0	(0, 5)	0.8 ± 0.5	(0, 2)
2008	<i>Phylctenactis tuberculosa</i>	-	-	-	-	0.3 ± 0.3	(0, 1)	-	-	-	-
2008	<i>Plagusia chabris</i>	1.3 ± 0.8	(0, 3)	0.8 ± 0.8	(0, 3)	2.3 ± 1.9	(0, 8)	3 ± 1.9	(0, 8)	0.8 ± 0.3	(0, 1)
2008	<i>Plaxiphora oblecta</i>	0.3 ± 0.3	(0, 1)	-	-	0.3 ± 0.3	(0, 1)	0.5 ± 0.3	(0, 1)	-	-
2008	<i>Scutus breviculus</i>	0.5 ± 0.5	(0, 2)	-	-	-	-	2 ± 0.7	(1, 4)	-	-
2008	<i>Stichaster australis</i>	-	-	0.3 ± 0.3	(0, 1)	0.3 ± 0.3	(0, 1)	0.3 ± 0.3	(0, 1)	-	-
2008	<i>Stichopus mollis</i>	-	-	3.8 ± 1.3	(1, 7)	3.0 ± 0.7	(1, 4)	1.5 ± 0.9	(0, 4)	0.3 ± 0.3	(0, 1)
2008	<i>Stignaster inflatus</i>	-	-	-	-	-	-	-	-	-	-
2008	<i>Sypharochiton pelliserpentis</i>	-	-	-	-	0 ± 0	(0, 0)	-	-	-	-
2008	<i>Trochus viridis</i>	-	-	-	-	-	-	-	-	-	-
2008	<i>Turbo smaragdus</i>	0.3 ± 0.3	(0, 1)	-	-	1.5 ± 0.9	(0, 3)	7.5 ± 2.9	(1, 15)	9.8 ± 5.6	(0, 25)
2009	<i>Aphelodoris luctosa</i>	-	-	0.5 ± 0.3	(0, 1)	0.5 ± 0.3	(0, 1)	0.5 ± 0.5	(0, 2)	1.3 ± 0.5	(0, 2)
2009	<i>Archidoris wellingtoniensis</i>	-	-	-	-	-	-	-	-	-	-
2009	<i>Argobuccinum pustulosum tumidum</i>	0.3 ± 0.3	(0, 1)	-	-	-	-	-	-	-	-
2009	<i>Astrastole scabrus</i>	0.3 ± 0.3	(0, 1)	1.3 ± 0.5	(0, 2)	1.0 ± 0	(1, 1)	2.8 ± 1.5	(0, 7)	3.3 ± 0.6	(2, 5)
2009	<i>Buccilinum linea</i>	0.3 ± 0.3	(0, 1)	0.5 ± 0.5	(0, 2)	0.3 ± 0.3	(0, 1)	0.5 ± 0.5	(0, 2)	1.0 ± 0.6	(0, 2)
2009	<i>Cabestana spengleri</i>	-	-	-	-	-	-	-	-	-	-
2009	<i>Cantharidus opalus</i>	-	(0, 1)	0.5 ± 0.3	(0, 1)	-	-	-	-	1.0 ± 0.6	(0, 2)
2009	<i>Cantharidus purpureus</i>	0.3 ± 0.3	-	0.5 ± 0.3	(0, 1)	0.5 ± 0.5	(0, 2)	0.3 ± 0.3	(0, 1)	0.8 ± 0.5	(0, 2)
2009	<i>Chromodoris aureomarginata</i>	0.3 ± 0.3	(0, 1)	-	-	-	-	0.3 ± 0.3	(0, 1)	-	-
2009	<i>Cominella virgata</i>	-	-	-	-	-	-	-	-	-	-

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Species	Princess Bay		Palmer Head		Flax Bay		Breaker Bay		Barret Reef	
	Mean, SE	Min, Max	Mean, SE	Min, Max	Mean, SE	Min, Max	Mean, SE	Min, Max	Mean, SE	Min, Max
<i>Pectinura maculata</i>	3 ± 2.3	(0, 10)	19.3 ± 1.9	(14, 23)	20 ± 3.8	(13, 27)			0 ± 0	(0, 0)
<i>Penion sulcatus</i>	-	-	-	-	-	-			-	-
<i>Pentagonaster pulchellus</i>	1.5 ± 0.6	(0, 3)	2.75 ± 0.25	(2, 3)	0.3 ± 0.3	(0, 1)			4.3 ± 1.4	(1, 7)
<i>Phylctenactis tuberculosa</i>	0.3 ± 0.3	(0, 1)	-	-	1.3 ± 0.5	(0, 2)			-	-
<i>Plagusia chabris</i>	0.5 ± 0.5	(0, 2)	0.5 ± 0.5	(0, 2)	-	-			1.8 ± 1.2	(0, 5)
<i>Plaxiphora obtecta</i>	0.5 ± 0.3	(0, 1)	2.3 ± 0.9	(1, 5)	-	-			0.3 ± 0.3	(0, 1)
<i>Scutus breviculus</i>	1.3 ± 0.5	(0, 2)	0.3 ± 0.3	(0, 1)	11.5 ± 4.2	(0, 19)			0 ± 0	(0, 0)
<i>Stichaster australis</i>	0.5 ± 0.3	(0, 1)	0.3 ± 0.3	(0, 1)	0.3 ± 0.3	(0, 1)			0.3 ± 0.3	(0, 1)
<i>Stichopus mollis</i>	1.3 ± 1.3	(0, 5)	0.5 ± 0.3	(0, 1)	14.8 ± 4.4	(8, 27)			6.3 ± 2.5	(0, 12)
<i>Stignaster inflatus</i>	-	-	-	-	-	-			0.3 ± 0.3	(0, 1)
<i>Sypharochiton pelliserpentis</i>	-	-	-	-	0.5 ± 0.5	(0, 2)			-	-
<i>Trochus viridis</i>	0 ± 0	(0, 0)	-	-	-	-			-	-
<i>Turbo smaragdus</i>	30.3 ± 8.4	(14, 54)	10.3 ± 3.8	(4, 19)	14.5 ± 6.7	(4, 32)			-	-
<i>Aphelodoris luctosa</i>	0.8 ± 0.8	(0, 3)	0.3 ± 0.3	(0, 1)	1.0 ± 0.4	(0, 2)	2.0 ± 0.9	(0, 4)	1.3 ± 0.9	(0, 4)
<i>Archidoris wellingtoniensis</i>	0.3 ± 0.3	(0, 1)	0.3 ± 0.3	(0, 1)	-	-	-	-	-	-
<i>Argobuccinum pustulosum</i>	-	-	-	-	-	-	0.3 ± 0.3	(0, 1)	0 ± 0	(0, 0)
<i>tumidum</i>										
<i>Astrostole scabrus</i>	5.0 ± 0.7	(4, 7)	1.5 ± 0.6	(0, 3)	1.8 ± 0.9	(0, 4)	3.3 ± 0.5	(2, 4)	2.8 ± 0.9	(1, 5)
<i>Buccilinum linea</i>	0.3 ± 0.3	(0, 1)	-	-	0.5 ± 0.3	(0, 1)	0.3 ± 0.3	(0, 1)	-	-
<i>Cabestana spengleri</i>	-	-	-	-	-	-	0.5 ± 0.5	(0, 2)	0.5 ± 0.5	(0, 2)
<i>Cantharidus opalus</i>	-	-	-	-	-	-	1.0 ± 0.7	(0, 3)	0.3 ± 0.3	(0, 1)
<i>Cantharidus purpureus</i>	0.5 ± 0.3	(0, 1)	-	-	0.3 ± 0.3	(0, 1)	0.3 ± 0.3	(0, 1)	0.3 ± 0.3	(0, 1)
<i>Chromodoris</i>	-	-	0.3 ± 0.3	(0, 1)	-	-	-	-	-	-
<i>aureomarginata</i>										
<i>Cominella virgata</i>	-	-	-	-	0.8 ± 0.3	(0, 1)	0.5 ± 0.5	(0, 2)	-	-

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Survey year	Species	2nd Wash		1st Wash		Red Rocks		Yung Pen		The Sirens	
		Mean, SE	Min, Max	Mean, SE	Min, Max	Mean, SE	Min, Max	Mean, SE	Min, Max	Mean, SE	Min, Max
2009	<i>Cookia sulcata</i>	0.3 ± 0.3	(0, 1)	0.8 ± 0.5	(0, 2)	0.8 ± 0.3	(0, 1)	1.0 ± 0.7	(0, 3)	0.8 ± 0.8	(0, 3)
2009	<i>Coscinasterias muricata</i>	-	-	-	-	-	-	-	-	-	-
2009	<i>Diplodontias dilatatus</i>	1.5 ± 0.3	(1, 2)	0.3 ± 0.3	(0, 1)	0.8 ± 0.5	(0, 2)	2.3 ± 0.9	(1, 5)	1.0 ± 0.4	(0, 2)
2009	<i>Evechinus chloroticus</i>	9.3 ± 0.8	(7, 11)	25.5 ± 7.1	(10, 42)	4.3 ± 1.1	(2, 7)	3.3 ± 1.1	(0, 5)	3.8 ± 2.8	(0, 12)
2009	<i>Haliotis australis</i>	1.3 ± 0.9	(0, 4)	-	-	1.3 ± 0.9	(0, 4)	5.3 ± 3.4	(0, 15)	7.0 ± 1.9	(2, 11)
2009	<i>Haliotis iris</i>	-	-	-	-	0.3 ± 0.3	(0, 1)	3.0 ± 1.5	(0, 7)	12.8 ± 7.4	(0, 26)
2009	<i>Haustrum haustorum</i>	-	-	-	-	0.8 ± 0.3	(0, 1)	-	-	-	-
2009	<i>Ischnochiton maorianus</i>	-	-	-	-	-	-	-	-	-	-
2009	<i>Jasus edwardsii</i>	0.8 ± 0.5	(0, 2)	0.5 ± 0.3	(0, 1)	10.0 ± 6.3	(1, 28)	0.5 ± 0.5	(0, 2)	7.8 ± 4.9	(0, 22)
2009	<i>Maoricolpus roseus</i>	-	-	0.3 ± 0.3	(0, 1)	-	-	-	-	-	-
2009	<i>Modelia granulosa</i>	0.3 ± 0.3	(0, 1)	0.8 ± 0.5	(0, 2)	0.75 ± 0.25	(0, 1)	-	-	-	-
2009	<i>Octopus vulgaris</i>	-	-	-	-	-	-	-	-	-	-
2009	<i>Patiriella regularis</i>	13.3 ± 4.4	(5, 25)	3 ± 1.4	(1, 7)	42.5 ± 9.0	(29, 69)	28.8 ± 7.8	(7, 44)	5.0 ± 1.9	(1, 10)
2009	<i>Pectinura maculata</i>	-	-	0.3 ± 0.3	(0, 1)	2.0 ± 0.4	(1, 3)	2.8 ± 1.1	(0, 5)	-	-
2009	<i>Penion sulcatus</i>	-	-	-	-	-	-	-	-	-	-
2009	<i>Pentagonaster pulchellus</i>	2 ± 0.8	(0, 4)	3.8 ± 1.1	(1, 6)	3.0 ± 0.6	(2, 4)	2.8 ± 1.2	(1, 6)	0.5 ± 0.3	(0, 1)
2009	<i>Phylctenactis tuberculosa</i>	-	-	0.3 ± 0.3	(0, 1)	-	-	0.3 ± 0.3	(0, 1)	0.5 ± 0.3	(0, 1)
2009	<i>Plagusia chabris</i>	0.5 ± 0.5	(0, 2)	0.5 ± 0.5	(0, 2)	1.8 ± 0.9	(0, 4)	2.0 ± 0.9	(0, 4)	3.8 ± 0.8	(2, 5)
2009	<i>Plaxiphora oblecta</i>	1 ± 0.7	(0, 3)	-	-	-	-	0.3 ± 0.3	(0, 1)	-	-
2009	<i>Scutus breviculus</i>	-	-	-	-	-	-	2.0 ± 1.7	(0, 7)	-	-
2009	<i>Stegnater inflatus</i>	-	-	-	-	-	-	0.3 ± 0.3	(0, 1)	-	-
2009	<i>Stichaster australis</i>	-	-	-	-	0.5 ± 0.3	(0, 1)	1.0 ± 0.7	(0, 3)	-	-
2009	<i>Stichopus mollis</i>	0.5 ± 0.5	(0, 2)	6.3 ± 1.3	(5, 10)	6.0 ± 4.0	(1, 18)	0.8 ± 0.5	(0, 2)	4.0 ± 2.1	(0, 10)
2009	<i>Sypharochiton pelliserpentis</i>	-	-	-	-	-	-	-	-	-	-
2009	<i>Trochus viridis</i>	-	-	-	-	-	-	-	-	-	-
2009	<i>Turbo smaragdus</i>	-	-	0.3 ± 0.3	(0, 1)	1.3 ± 1.3	(0, 5)	4.3 ± 1.1	(2, 7)	19.3 ± 10.5	(1, 47)

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Species	Princess Bay		Palmer Head		Flax Bay		Breaker Bay		Barret Reef	
	Mean, SE	Min, Max	Mean, SE	Min, Max	Mean, SE	Min, Max	Mean, SE	Min, Max	Mean, SE	Min, Max
<i>Cookia sulcata</i>	1.3 ± 1.3	(0, 5)	-	-	1.8 ± 0.8	(0, 3)	-	-	0.3 ± 0.3	(0, 1)
<i>Coscinasterias muricata</i>	-	-	-	-	1.0 ± 0.6	(0, 2)	-	-	0.3 ± 0.3	(0, 1)
<i>Diplodontias dilatatus</i>	-	-	2.5 ± 0.9	(1, 5)	-	-	2.8 ± 0.5	(2, 4)	2.5 ± 1.0	(0, 5)
<i>Evechinus chloroticus</i>	2.3 ± 0.8	(1, 4)	6.8 ± 3.1	(3, 16)	3.5 ± 1.5	(1, 7)	4.8 ± 2.68	(0, 12)	8 ± 2.3	(3, 14)
<i>Haliotis australis</i>	20.3 ± 3.6	(14, 30)	0.5 ± 0.3	(0, 1)	4.8 ± 1.7	(1, 9)	9 ± 3.8	(0, 17)	1.5 ± 0.3	(1, 2)
<i>Haliotis iris</i>	12.0 ± 4.2	(4, 22)	4.5 ± 3.1	(0, 13)	22.5 ± 3.4	(16, 31)	0.8 ± 0.8	(0, 3)	14.5 ± 6.6	(2, 32)
<i>Haustrum haustorum</i>	1.0 ± 0.4	(0, 2)	-	-	0.5 ± 0.5	(0, 2)	-	-	0 ± 0	(0, 0)
<i>Ischnochiton maorianus</i>	-	-	-	-	-	-	-	-	-	-
<i>Jasus edwardsii</i>	1.3 ± 0.9	(0, 4)	2.0 ± 1.1	(0, 5)	0.5 ± 0.3	(0, 1)	-	-	2.3 ± 0.9	(0, 4)
<i>Maoricolpus roseus</i>	-	-	0.3 ± 0.3	(0, 1)	-	-	0.3 ± 0.3	(0, 1)	-	-
<i>Modelia granulosa</i>	-	-	-	-	-	-	-	-	-	-
<i>Octopus vulgaris</i>	-	-	-	-	0.3 ± 0.3	(0, 1)	-	-	0.3 ± 0.3	(0, 1)
<i>Patiriella regularis</i>	65.8 ± 10.3	(37, 81)	45.8 ± 11.6	(21, 71)	116.3 ± 12.1	(81, 137)	49.5 ± 24.3	(6, 108)	5.0 ± 3.1	(0, 14)
<i>Pectinura maculata</i>	1 ± 1	(0, 4)	12.8 ± 6.2	(0, 27)	29.3 ± 11.3	(6, 60)	47.5 ± 24.3	(2, 93)	-	-
<i>Penion sulcatus</i>	-	-	-	-	-	-	-	-	0.3 ± 0.3	(0, 1)
<i>Pentagonaster pulchellus</i>	0.8 ± 0.5	(0, 2)	3.5 ± 1.5	(1, 7)	0.3 ± 0.3	(0, 1)	10.0 ± 2.4	(6, 16)	3.8 ± 0.9	(2, 6)
<i>Phylctenactis tuberculosa</i>	1.3 ± 0.8	(0, 3)	-	-	0.3 ± 0.3	(0, 1)	-	-	-	-
<i>Plagusia chabris</i>	4.0 ± 0.8	(2, 6)	0.8 ± 0.5	(0, 2)	0.5 ± 0.5	(0, 2)	1.5 ± 1.5	(0, 6)	1.5 ± 0.5	(0, 2)
<i>Plaxiphora oblecta</i>	1 ± 1	(0, 4)	0.3 ± 0.3	(0, 1)	0.3 ± 0.3	(0, 1)	0.3 ± 0.3	(0, 1)	-	-
<i>Scutus breviculus</i>	1.3 ± 0.8	(0, 3)	-	-	13.3 ± 7.0	(2, 32)	15.0 ± 4.1	(7, 23)	-	-
<i>Stegnater inflatus</i>	0.5 ± 0.5	(0, 2)	0.8 ± 0.5	(0, 2)	-	-	1.8 ± 0.9	(0, 4)	0.5 ± 0.3	(0, 1)
<i>Stichaster australis</i>	-	-	0.3 ± 0.3	(0, 1)	-	-	-	-	0 ± 0	(0, 0)
<i>Stichopus mollis</i>	0.8 ± 0.5	(0, 2)	2.3 ± 1.0	(0, 4)	30 ± 4.3	(22, 42)	2.8 ± 2.1	(0, 9)	7.3 ± 3.1	(2, 16)
<i>Sypharochiton pelliserpentis</i>	-	-	-	-	-	-	-	-	-	-
<i>Trochus viridis</i>	-	-	-	-	0.5 ± 0.3	(0, 1)	-	-	0.5 ± 0.3	(0, 1)
<i>Turbo smaragdus</i>	16.8 ± 5.2	(2, 26)	1.5 ± 0.9	(0, 4)	47.0 ± 2.0	(41, 50)	-	-	-	-

Appendix F (chapter 3). Mean abundance of **sessile macro-invertebrates** (percent cover $m^{-2} \pm SE$, minimum (min) and maximum (max)) assessed for two annual baseline surveys of Wellington south coast's rocky-reef epibenthic communities (5-12m depth). This epibenthic group includes Ascidia, Bryozoa, Hydrozoa, and Porifera. Surveys were conducted during summer at nine sites in 2008 and ten sites in 2009 (Breaker Bay added). Organisms from these groups were labelled as morphotypes. Sites are listed here from west (2nd Wash) to east (Barret Reef).

Survey year	Species	2nd Wash		1st Wash		Red Rocks		The Sirens		Princess Bay	
		Mean, SE	Min, Max	Mean, SE	Min, Max	Mean, SE	Min, Max	Mean, SE	Min, Max	Mean, SE	Min, Max
2008	Ascidia A	0.05 ± 0.05	(0, 0.5)	0.1 ± 0.1	(0, 1)	-	-	-	-	-	-
2008	Ascidia B	0.05 ± 0.05	(0, 0.5)	-	-	-	-	-	-	-	-
2008	Ascidia C	-	-	-	-	-	-	-	-	-	-
2008	Ascidia D	0.15 ± 0.07	(0, 0.5)	0.3 ± 0.15	(0, 1)	0.4 ± 0.16	(0, 1)	-	-	-	-
2008	Bryozoa A	-	-	-	-	-	-	-	-	-	-
2008	Bryozoa B	-	-	-	-	-	-	-	-	-	-
2008	Bryozoa C	-	-	-	-	-	-	-	-	-	-
2008	Bryozoa D	-	-	0.05 ± 0.05	(0, 0.5)	-	-	-	-	0.1 ± 0.1	(0, 1)
2008	Bryozoa E	-	-	0.15 ± 0.10	(0, 1)	0.1 ± 0.1	(0, 1)	-	-	-	-
2008	Bryozoa F	-	-	-	-	-	-	-	-	-	-
2008	Bryozoa G	0.55 ± 0.39	(0, 4)	0.6 ± 0.20	(0, 2)	-	-	0.2 ± 0.13	(0, 1)	-	-
2008	Bryozoa H	-	-	-	-	-	-	-	-	-	-
2008	Hydrozoa A	0.2 ± 0.08	(0, 0.5)	0.3 ± 0.21	(0, 2)	-	-	-	-	-	-
2008	Hydrozoa B	0.1 ± 0.1	(0, 1)	0.1 ± 0.1	(0, 1)	0.6 ± 0.16	(0, 1)	-	-	-	-
2008	Porifera A	-	-	0.05 ± 0.05	(0, 0.5)	0.3 ± 0.15	(0, 1)	0.2 ± 0.13	(0, 1)	0.1 ± 0.1	(0, 1)
2008	Porifera B	-	-	-	-	0.5 ± 0.16	(0, 1)	-	-	0.1 ± 0.1	(0, 1)
2008	Porifera C	-	-	-	-	-	-	-	-	-	-
2008	Porifera D	0.05 ± 0.05	(0, 0.5)	-	-	-	-	-	-	-	-
2008	Porifera E	0.1 ± 0.06	(0, 0.5)	-	-	0.2 ± 0.13	(0, 1)	0.1 ± 0.1	(0, 1)	0.1 ± 0.1	(0, 1)
2008	Porifera F	-	-	-	-	-	-	0.6 ± 0.49	(0, 5)	-	-
2008	Porifera G	0.2 ± 0.08	(0, 0.5)	0.1 ± 0.1	(0, 1)	0.1 ± 0.1	(0, 1)	0.05 ± 0.05	(0, 0.5)	-	-
2008	Porifera H	-	-	-	-	-	-	-	-	-	-
2008	Porifera I	0.05 ± 0.05	(0, 0.5)	0.05 ± 0.05	(0, 0.5)	0.2 ± 0.13	(0, 1)	-	-	0.1 ± 0.1	(0, 1)
2008	Porifera J	-	-	-	-	-	-	-	-	-	-
2008	Porifera K	-	-	-	-	-	-	-	-	-	-
2008	Porifera L	0.45 ± 0.39	(0, 4)	-	-	0.2 ± 0.13	(0, 1)	0.25 ± 0.13	(0, 1)	-	-
2008	Porifera M	-	-	-	-	0.8 ± 0.38	(0, 4)	-	-	-	-
2008	Porifera N	0.1 ± 0.06	(0, 0.5)	0.1 ± 0.1	(0, 1)	-	-	-	-	-	-
2008	Porifera O	0.05 ± 0.05	(0, 0.5)	-	-	-	-	-	-	-	-
2008	Porifera P	-	-	0.05 ± 0.05	(0, 0.5)	0.45 ± 0.15	(0, 1)	0.1 ± 0.1	(0, 1)	-	-

Appendix F (chapter 3). Mean abundance of sessile macro-invertebrates, page 2.

Survey year	Species	Yung Pen		Flax Bay		Palmer Head		Breaker Bay		Barret Reef		N
		Mean, SE	Min, Max	Mean, SE	Min, Max	Mean, SE	Min, Max	Mean, SE	Min, Max	Mean, SE	Min, Max	
2008	Ascidia A	0.1 ± 0.1	(0, 1)	-	-	-	-	-	-	0.2 ± 0.13	(0, 1)	10
2008	Ascidia B	-	-	-	-	0.1 ± 0.1	(0, 1)	-	-	0.4 ± 0.22	(0, 2)	10
2008	Ascidia C	-	-	-	-	-	-	-	-	0.7 ± 0.59	(0, 6)	10
2008	Ascidia D	0.2 ± 0.13	(0, 1)	-	-	-	-	-	-	0.2 ± 0.13	(0, 1)	10
2008	Bryozoa A	-	-	0.4 ± 0.4	(0, 4)	-	-	-	-	0.5 ± 0.30	(0, 3)	10
2008	Bryozoa B	-	-	-	-	-	-	-	-	0.25 ± 0.13	(0, 1)	10
2008	Bryozoa C	-	-	-	-	-	-	-	-	0.1 ± 0.1	(0, 1)	10
2008	Bryozoa D	0.1 ± 0.1	(0, 1)	-	-	-	-	-	-	-	-	10
2008	Bryozoa E	0.4 ± 0.4	(0, 4)	-	-	-	-	-	-	0.1 ± 0.1	(0, 1)	10
2008	Bryozoa F	-	-	-	-	-	-	-	-	0.05 ± 0.05	(0, 0.5)	10
2008	Bryozoa G	-	-	-	-	0.3 ± 0.15	(0, 1)	-	-	0.4 ± 0.16	(0, 1)	10
2008	Bryozoa H	-	-	-	-	-	-	-	-	-	-	10
2008	Hydrozoa A	-	-	-	-	-	-	-	-	0.4 ± 0.4	(0, 4)	10
2008	Hydrozoa B	0.3 ± 0.15	(0, 1)	-	-	-	-	-	-	0.2 ± 0.13	(0, 1)	10
2008	Porifera A	0.1 ± 0.1	(0, 1)	-	-	-	-	-	-	0.2 ± 0.13	(0, 1)	10
2008	Porifera B	0.4 ± 0.16	(0, 1)	-	-	-	-	-	-	0.1 ± 0.1	(0, 1)	10
2008	Porifera C	-	-	0.1 ± 0.1	(0, 1)	-	-	-	-	-	-	10
2008	Porifera D	-	-	-	-	-	-	-	-	0.4 ± 0.16	(0, 1)	10
2008	Porifera E	-	-	-	-	0.1 ± 0.1	(0, 1)	-	-	-	-	10
2008	Porifera F	-	-	-	-	-	-	-	-	-	-	10
2008	Porifera G	-	-	-	-	-	-	-	-	0.1 ± 0.1	(0, 1)	10
2008	Porifera H	-	-	-	-	-	-	-	-	0.3 ± 0.15	(0, 1)	10
2008	Porifera I	-	-	-	-	-	-	-	-	-	-	10
2008	Porifera J	0.1 ± 0.1	(0, 1)	-	-	-	-	-	-	-	-	10
2008	Porifera K	-	-	-	-	-	-	-	-	0.1 ± 0.1	(0, 1)	10
2008	Porifera L	-	-	0.1 ± 0.1	(0, 1)	-	-	-	-	0.1 ± 0.1	(0, 1)	10
2008	Porifera M	-	-	-	-	-	-	-	-	-	-	10
2008	Porifera N	-	-	-	-	-	-	-	-	-	-	10
2008	Porifera O	0.1 ± 0.1	(0, 1)	-	-	-	-	-	-	-	-	10
2008	Porifera P	-	-	-	-	-	-	-	-	-	-	10

Appendix F (chapter 3). Mean abundance of sessile macro-invertebrates, page 3.

Survey year	Species	2nd Wash		1st Wash		Red Rocks		The Sirens		Princess Bay	
		Mean, SE	Min, Max	Mean, SE	Min, Max	Mean, SE	Min, Max	Mean, SE	Min, Max	Mean, SE	Min, Max
2009	Ascidia A	0.09 ± 0.04	(0, 0.5)	0.12 ± 0.09	(0, 1)	0.01 ± 0.01	(0, 0.1)	0.3 ± 0.3	(0, 3)	0.01 ± 0.01	(0, 0.1)
2009	Ascidia B	0.02 ± 0.01	(0, 0.1)	0.03 ± 0.01	(0, 0.1)	0.03 ± 0.01	(0, 0.1)	-	-	-	-
2009	Ascidia C	-	-	-	-	-	-	-	-	-	-
2009	Ascidia D	0.03 ± 0.01	(0, 0.1)	0.05 ± 0.01	(0, 0.1)	0.05 ± 0.01	(0, 0.1)	0.05 ± 0.05	(0, 0.5)	0.02 ± 0.01	(0, 0.1)
2009	Bryozoa A	-	-	-	-	-	-	-	-	-	-
2009	Bryozoa B	-	-	-	-	-	-	-	-	-	-
2009	Bryozoa C	-	-	-	-	-	-	-	-	-	-
2009	Bryozoa D	-	-	-	-	-	-	-	-	-	-
2009	Bryozoa E	-	-	0.1 ± 0.1	(0, 1)	-	-	-	-	-	-
2009	Bryozoa F	0.01 ± 0.01	(0, 0.1)	0.06 ± 0.04	(0, 0.5)	-	-	-	-	-	-
2009	Bryozoa G	0.4 ± 0.22	(0, 2)	0.4 ± 0.20	(0, 2)	0.2 ± 0.13	(0, 1)	-	-	0.02 ± 0.01	(0, 0.1)
2009	Bryozoa H	-	-	-	-	-	-	-	-	-	-
2009	Hydrozoa A	0.3 ± 0.13	(0, 1)	0.65 ± 0.23	(0, 2)	0.15 ± 0.07	(0, 0.5)	-	-	-	-
2009	Hydrozoa B	0.3 ± 0.15	(0, 1)	0.1 ± 0.1	(0, 1)	0.11 ± 0.06	(0, 0.5)	0.15 ± 0.07	(0, 0.5)	0.08 ± 0.04	(0, 0.5)
2009	Porifera A	0.1 ± 0.1	(0, 1)	-	-	-	-	-	-	-	-
2009	Porifera B	0.25 ± 0.13	(0, 1)	0.2 ± 0.13	(0, 1)	-	-	0.2 ± 0.11	(0, 1)	0.25 ± 0.08	(0, 0.5)
2009	Porifera C	0.02 ± 0.01	(0, 0.1)	0.4 ± 0.16	(0, 1)	0.01 ± 0.01	(0, 0.1)	0.01 ± 0.01	(0, 0.1)	-	-
2009	Porifera D	0.3 ± 0.21	(0, 2)	0.1 ± 0.1	(0, 1)	-	-	-	-	-	-
2009	Porifera E	0.13 ± 0.09	(0, 1)	-	-	0.16 ± 0.07	(0, 0.5)	0.21 ± 0.13	(0, 1)	0.08 ± 0.04	(0, 0.5)
2009	Porifera F	0.05 ± 0.05	(0, 0.5)	0.1 ± 0.1	(0, 1)	-	-	-	-	0.2 ± 0.2	(0, 2)
2009	Porifera G	0.02 ± 0.01	(0, 0.1)	-	-	0.01 ± 0.01	(0, 0.1)	0.1 ± 0.06	(0, 0.5)	0.01 ± 0.01	(0, 0.1)
2009	Porifera H	-	-	0.1 ± 0.1	(0, 1)	-	-	-	-	-	-
2009	Porifera I	-	-	0.15 ± 0.10	(0, 1)	0.18 ± 0.10	(0, 1)	0.02 ± 0.01	(0, 0.1)	-	-
2009	Porifera J	-	-	-	-	-	-	-	-	-	-
2009	Porifera K	-	-	-	-	-	-	-	-	-	-
2009	Porifera L	-	-	-	-	-	-	-	-	-	-
2009	Porifera M	0.02 ± 0.01	(0, 0.1)	0.01 ± 0.01	(0, 0.1)	0.06 ± 0.01	(0, 0.1)	0.03 ± 0.01	(0, 0.1)	0.05 ± 0.05	(0, 0.5)
2009	Porifera N	-	-	-	-	-	-	-	-	-	-
2009	Porifera O	-	-	-	-	-	-	-	-	-	-
2009	Porifera P	0.11 ± 0.09	(0, 1)	-	-	0.36 ± 0.14	(0, 1)	-	-	0.11 ± 0.04	(0, 0.5)

Appendix F (chapter 3). Mean abundance of sessile macro-invertebrates, final.

Survey year	Species	Yung Pen		Flax Bay		Palmer Head		Breaker Bay		Barret Reef		N
		Mean, SE	Min, Max	Mean, SE	Min, Max	Mean, SE	Min, Max	Mean, SE	Min, Max	Mean, SE	Min, Max	
2009	Ascidia A	0.02 ± 0.01	(0, 0.1)	0.05 ± 0.05	(0, 0.5)	0.2 ± 0.11	(0, 1)	0.1 ± 0.05	(0, 0.5)	0.07 ± 0.04	(0, 0.5)	10
2009	Ascidia B	-	-	-	-	0.05 ± 0.05	(0, 0.5)	-	-	0.02 ± 0.01	(0, 0.1)	10
2009	Ascidia C	-	-	-	-	0.1 ± 0.06	(0, 0.5)	-	-	0.36 ± 0.23	(0, 2)	10
2009	Ascidia D	0.17 ± 0.10	(0, 1)	0.05 ± 0.05	(0, 0.5)	0.03 ± 0.01	(0, 0.1)	0.2 ± 0.13	(0, 1)	0.23 ± 0.14	(0, 1.5)	10
2009	Bryozoa A	-	-	-	-	-	-	-	-	0.04 ± 0.01	(0, 0.1)	10
2009	Bryozoa B	-	-	-	-	0.05 ± 0.05	(0, 0.5)	0.15 ± 0.10	(0, 1)	0.56 ± 0.29	(0, 3)	10
2009	Bryozoa C	-	-	-	-	-	-	-	-	-	-	10
2009	Bryozoa D	-	-	-	-	-	-	-	-	-	-	10
2009	Bryozoa E	-	-	-	-	0.15 ± 0.10	(0, 1)	-	-	0.07 ± 0.04	(0, 0.5)	10
2009	Bryozoa F	-	-	-	-	-	-	-	-	0.06 ± 0.04	(0, 0.5)	10
2009	Bryozoa G	-	-	0.1 ± 0.1	(0, 1)	0.2 ± 0.11	(0, 1)	-	-	0.11 ± 0.06	(0, 0.5)	10
2009	Bryozoa H	-	-	-	-	-	-	-	-	0.05 ± 0.05	(0, 0.5)	10
2009	Hydrozoa A	-	-	-	-	0.2 ± 0.11	(0, 1)	-	-	0.05 ± 0.05	(0, 0.5)	10
2009	Hydrozoa B	0.02 ± 0.01	(0, 0.1)	-	-	-	-	-	-	-	-	10
2009	Porifera A	-	-	-	-	-	-	-	-	0.01 ± 0.01	(0, 0.1)	10
2009	Porifera B	0.2 ± 0.11	(0, 1)	-	-	-	-	-	-	-	-	10
2009	Porifera C	-	-	-	-	0.01 ± 0.01	(0, 0.1)	-	-	-	-	10
2009	Porifera D	-	-	-	-	0.15 ± 0.15	(0, 1.5)	-	-	0.15 ± 0.15	(0, 1.5)	10
2009	Porifera E	0.01 ± 0.01	(0, 0.1)	0.1 ± 0.1	(0, 1)	0.1 ± 0.06	(0, 0.5)	0.1 ± 0.1	(0, 1)	0.02 ± 0.01	(0, 0.1)	10
2009	Porifera F	-	-	-	-	0.5 ± 0.5	(0, 5)	-	-	0.5 ± 0.5	(0, 5)	10
2009	Porifera G	-	-	-	-	-	-	-	-	-	-	10
2009	Porifera H	-	-	-	-	-	-	-	-	-	-	10
2009	Porifera I	-	-	-	-	-	-	-	-	-	-	10
2009	Porifera J	-	-	-	-	-	-	-	-	-	-	10
2009	Porifera K	-	-	-	-	-	-	-	-	-	-	10
2009	Porifera L	-	-	0.01 ± 0.01	(0, 0.1)	0.21 ± 0.13	(0, 1)	-	-	-	-	10
2009	Porifera M	0.07 ± 0.04	(0, 0.5)	-	-	0.01 ± 0.01	(0, 0.1)	-	-	-	-	10
2009	Porifera N	-	-	-	-	-	-	-	-	-	-	10
2009	Porifera O	-	-	-	-	-	-	-	-	-	-	10
2009	Porifera P	0.01 ± 0.01	(0, 0.1)	-	-	-	-	0.1 ± 0.1	(0, 1)	0.05 ± 0.05	(0, 0.5)	10

Appendix G (chapter 3). PERMANOVA pairwise comparisons for the significant interaction term “Year x Site” on macro-algal count data. The test used a subset of 11 species from Wellington south coast’s nearshore rocky-reef zone. The site “Breaker Bay” was only surveyed during the second year (2009). **Bolded** values are **not** significant at $p < 0.05$. p-values were derived from 9999 permutations.

Macro-algae: counts m ⁻² (subset of 11 species)							
Groups	t	p-value	Unique permutation	Groups	t	p-value	Unique permutation
Sites within 2008				Sites within 2009			
2nd Wash, 1st Wash	2.4523	0.0012	9444	2nd Wash, 1st Wash	1.9250	0.0106	9427
2nd Wash, Red Rocks	2.8176	0.0001	9166	2nd Wash, Red Rocks	1.8924	0.0050	9119
2nd Wash, Yung Pen	2.6193	0.0001	9417	2nd Wash, Yung Pen	2.1596	0.0005	9458
2nd Wash, The Sirens	2.1457	0.0032	9144	2nd Wash, The Sirens	2.0645	0.0140	9433
2nd Wash, Princess Bay	2.4313	0.0010	8714	2nd Wash, Princess Bay	2.3288	0.0019	9366
2nd Wash, Palmer Head	2.4075	0.0002	9192	2nd Wash, Palmer Head	2.5437	0.0012	9150
2nd Wash, Flax Bay	3.4926	0.0001	9172	2nd Wash, Flax Bay	4.0953	0.0002	9453
2nd Wash, Barret Reef	1.8976	0.0087	9450	2nd Wash, Barret Reef	2.9293	0.0007	9462
1st Wash, Red Rocks	4.0621	0.0001	9204	2nd Wash, Breaker Bay	3.9479	0.0001	9426
1st Wash, Yung Pen	3.2317	0.0001	9425	1st Wash, Red Rocks	1.1740	0.2521	9156
1st Wash, The Sirens	2.0600	0.0035	9196	1st Wash, Yung Pen	1.9059	0.0044	9427
1st Wash, Princess Bay	2.9217	0.0001	8672	1st Wash, The Sirens	2.0033	0.0079	9438
1st Wash, Palmer Head	1.0885	0.3050	9482	1st Wash, Princess Bay	1.9073	0.0132	9425
1st Wash, Flax Bay	3.2536	0.0001	9165	1st Wash, Palmer Head	1.7252	0.0247	9150
1st Wash, Barret Reef	1.3594	0.1410	9465	1st Wash, Flax Bay	2.4867	0.0001	9423
Red Rocks, Yung Pen	1.8433	0.0028	9159	1st Wash, Barret Reef	2.0254	0.0033	9459
Red Rocks, The Sirens	2.4760	0.0003	8842	1st Wash, Breaker Bay	3.0568	0.0001	9453
Red Rocks, Princess Bay	1.8101	0.0202	8152	Red Rocks, Yung Pen	1.4455	0.0750	9138
Red Rocks, Palmer Head	3.7798	0.0001	9206	Red Rocks, The Sirens	1.3666	0.1367	9180
Red Rocks, Flax Bay	3.7210	0.0001	8815	Red Rocks, Princess Bay	1.1365	0.2978	9185
Red Rocks, Barret Reef	3.7799	0.0001	9221	Red Rocks, Palmer Head	2.5536	0.0003	8816
Yung Pen, The Sirens	2.1911	0.0023	9145	Red Rocks, Flax Bay	2.4993	0.0001	9132
Yung Pen, Princess Bay	2.0277	0.0138	8713	Red Rocks, Barret Reef	2.9261	0.0001	9168
Yung Pen, Palmer Head	2.9532	0.0001	9433	Red Rocks, Breaker Bay	2.7402	0.0001	9165
Yung Pen, Flax Bay	3.0544	0.0001	9201	Yung Pen, The Sirens	1.3121	0.1531	9443
Yung Pen, Barret Reef	3.0930	0.0001	9426	Yung Pen, Princess Bay	1.1073	0.3161	9476
The Sirens, Princess Bay	1.0983	0.3190	6661	Yung Pen, Palmer Head	2.9320	0.0001	9225
The Sirens, Palmer Head	1.6952	0.0321	8728	Yung Pen, Flax Bay	2.8287	0.0001	9452
The Sirens, Flax Bay	2.3462	0.0012	7927	Yung Pen, Barret Reef	3.2061	0.0003	9406
The Sirens, Barret Reef	2.4526	0.0003	9169	Yung Pen, Breaker Bay	2.7936	0.0003	9408
Princess Bay, Palmer Head	2.4390	0.0010	7917	The Sirens, Princess Bay	1.0865	0.3350	9422
Princess Bay, Flax Bay	2.6545	0.0010	6627	The Sirens, Palmer Head	3.4118	0.0008	9216
Princess Bay, Barret Reef	3.0248	0.0001	8681	The Sirens, Flax Bay	4.0973	0.0001	9429
Palmer Head, Flax Bay	2.3625	0.0005	8683	The Sirens, Barret Reef	4.3188	0.0001	9424
Palmer Head, Barret Reef	1.7883	0.0069	9429	The Sirens, Breaker Bay	4.2547	0.0001	9455
Flax Bay, Barret Reef	3.5565	0.0001	9232	Princess Bay, Palmer Head	2.8009	0.0019	9146
				Princess Bay, Flax Bay	2.9736	0.0002	9407
				Princess Bay, Barret Reef	3.4491	0.0001	9439
				Princess Bay, Breaker Bay	2.7199	0.0005	9463
				Palmer Head, Flax Bay	3.7074	0.0001	9186
				Palmer Head, Barret Reef	2.1752	0.0046	9202
				Palmer Head, Breaker Bay	2.8940	0.0002	9226
				Flax Bay, Barret Reef	2.9325	0.0003	9429
				Flax Bay, Breaker Bay	3.2777	0.0002	9445
				Barret Reef, Breaker Bay	3.1669	0.0001	9439

Appendix H (chapter 3). PERMANOVA pairwise comparisons for the significant interaction term “Year x Site” on mobile macro-invertebrate count data. Data were collected during two annual baseline surveys of Wellington’s south coast nearshore rocky-reefs. The site “Breaker Bay” was only surveyed during the second year (2009). **Bolded** values are **not** significant at $p < 0.05$. Actual p-values were derived from 9999 permutations.

Mobile macro-invertebrates: count m ⁻²							
Groups	t	p-value	Unique permutation	Groups	t	p-value	Unique permutation
Sites within 2008				Sites within 2009			
2nd Wash, 1st Wash	1.5066	0.0566	35	2nd Wash, 1st Wash	1.7249	0.0285	35
2nd Wash, Red Rocks	1.2831	0.2303	35	2nd Wash, Red Rocks	1.7969	0.0271	35
2nd Wash, Yung Pen	1.3527	0.0235	35	2nd Wash, Yung Pen	1.4893	0.0281	35
2nd Wash, The Sirens	1.6457	0.0836	35	2nd Wash, The Sirens	2.2144	0.0300	35
2nd Wash, Princess Bay	1.8295	0.0316	35	2nd Wash, Princess Bay	2.8852	0.0297	35
2nd Wash, Flax Bay	2.5462	0.0280	35	2nd Wash, Flax Bay	3.4729	0.0271	35
2nd Wash, Palmer Head	1.8640	0.0564	35	2nd Wash, Palmer Head	1.3210	0.1716	35
2nd Wash, Barret Reef	1.6620	0.0271	35	2nd Wash, Barret Reef	1.7781	0.0313	35
1st Wash, Red Rocks	1.5392	0.0571	35	2nd Wash, Breaker Bay	2.2180	0.0291	35
1st Wash, Yung Pen	2.3289	0.0291	35	1st Wash, Red Rocks	1.8229	0.0272	35
1st Wash, The Sirens	2.0243	0.0274	35	1st Wash, Yung Pen	2.0467	0.0302	35
1st Wash, Princess Bay	2.7531	0.0310	35	1st Wash, The Sirens	2.1372	0.0245	35
1st Wash, Flax Bay	3.0285	0.0283	35	1st Wash, Princess Bay	3.1801	0.0283	35
1st Wash, Palmer Head	2.9549	0.0287	35	1st Wash, Flax Bay	3.4824	0.0301	35
1st Wash, Barret Reef	1.6959	0.0277	35	1st Wash, Palmer Head	1.8799	0.0286	35
Red Rocks, Yung Pen	1.2728	0.2538	35	1st Wash, Barret Reef	1.7042	0.0558	35
Red Rocks, The Sirens	1.4683	0.1088	35	1st Wash, Breaker Bay	2.4411	0.0277	35
Red Rocks, Princess Bay	1.6906	0.0257	35	Red Rocks, Yung Pen	1.4126	0.0264	35
Red Rocks, Flax Bay	1.7368	0.0275	35	Red Rocks, The Sirens	2.1195	0.0280	35
Red Rocks, Palmer Head	1.7002	0.0306	35	Red Rocks, Princess Bay	2.5359	0.0285	35
Red Rocks, Barret Reef	1.5856	0.0288	35	Red Rocks, Flax Bay	3.2156	0.0285	35
Yung Pen, The Sirens	1.4363	0.1161	35	Red Rocks, Palmer Head	1.3394	0.0801	35
Yung Pen, Princess Bay	1.1047	0.2849	35	Red Rocks, Barret Reef	1.8827	0.0287	35
Yung Pen, Flax Bay	1.9416	0.0331	35	Red Rocks, Breaker Bay	2.4448	0.0286	35
Yung Pen, Palmer Head	1.7037	0.0273	35	Yung Pen, The Sirens	1.3944	0.0592	35
Yung Pen, Barret Reef	1.8016	0.0283	35	Yung Pen, Princess Bay	1.6299	0.0552	35
The Sirens, Princess Bay	1.5220	0.0576	35	Yung Pen, Flax Bay	2.3773	0.0282	35
The Sirens, Flax Bay	2.3398	0.0286	35	Yung Pen, Palmer Head	0.7004	0.9411	35
The Sirens, Palmer Head	2.0117	0.0293	35	Yung Pen, Barret Reef	1.4627	0.0315	35
The Sirens, Barret Reef	1.8363	0.0313	35	Yung Pen, Breaker Bay	1.4901	0.0269	35
Princess Bay, Flax Bay	2.0493	0.0278	35	The Sirens, Princess Bay	1.5802	0.0544	35
Princess Bay, Palmer Head	1.7367	0.0310	35	The Sirens, Flax Bay	2.5895	0.0303	35
Princess Bay, Barret Reef	2.1355	0.0309	35	The Sirens, Palmer Head	1.8742	0.0277	35
Flax Bay, Palmer Head	2.2394	0.0273	35	The Sirens, Barret Reef	1.5399	0.0276	35
Flax Bay, Barret Reef	2.7252	0.0305	35	The Sirens, Breaker Bay	2.2466	0.0311	35
Palmer Head, Barret Reef	2.1305	0.0291	35	Princess Bay, Flax Bay	2.5088	0.0282	35
				Princess Bay, Palmer Head	2.1996	0.0301	35
				Princess Bay, Barret Reef	2.4525	0.0280	35
				Princess Bay, Breaker Bay	2.6100	0.0298	35
				Flax Bay, Palmer Head	2.6562	0.0274	35
				Flax Bay, Barret Reef	2.9647	0.0282	35
				Flax Bay, Breaker Bay	2.8808	0.0297	35
				Palmer Head, Barret Reef	1.3198	0.1153	35
				Palmer Head, Breaker Bay	1.5490	0.0313	35
				Barret Reef, Breaker Bay	1.9416	0.0289	35

Appendix I (chapter 3). PERMANOVA pairwise comparisons for the significant interaction term “Year x Site” on sessile macro-invertebrate percent cover data. Data were collected during two annual baseline surveys of Wellington’s south coast nearshore rocky-reefs. The site “Breaker Bay” was only surveyed during the second year (2009). **Bolded** values are **not** significant at $p < 0.05$. p-values were derived from 9999 permutations.

Sessile macro-invertebrates: percent cover m ⁻² (all species surveyed)							
Groups	t	p-value	Unique permutation	Groups	t	p-value	Unique permutation
Sites within 2008				Sites within 2009			
2nd Wash, 1st Wash	1.0323	0.3697	6607	2nd Wash, 1st Wash	1.0293	0.3770	9461
2nd Wash, Red Rocks	2.2045	0.0005	8649	2nd Wash, Red Rocks	1.2955	0.1318	9176
2nd Wash, Yung Pen	1.5509	0.0231	4018	2nd Wash, Yung Pen	1.8285	0.0059	8671
2nd Wash, The Sirens	1.3542	0.0918	3208	2nd Wash, The Sirens	1.3496	0.0827	8665
2nd Wash, Princess Bay	1.8401	0.0044	981	2nd Wash, Princess Bay	1.6088	0.0184	9427
2nd Wash, Palmer Head	1.7062	0.0170	1399	2nd Wash, Palmer Head	1.1915	0.1785	7906
2nd Wash, Flax Bay	2.0835	0.0041	254	2nd Wash, Flax Bay	2.1145	0.0023	4970
2nd Wash, Barret Reef	1.5609	0.0071	9154	2nd Wash, Barret Reef	1.5823	0.0025	9394
1st Wash, Red Rocks	2.2700	0.0003	7908	2nd Wash, Breaker Bay	1.9283	0.0028	6616
1st Wash, Yung Pen	1.5396	0.0464	2511	1st Wash, Red Rocks	1.6957	0.0103	8745
1st Wash, The Sirens	1.4257	0.0780	1859	1st Wash, Yung Pen	2.2395	0.0010	7122
1st Wash, Princess Bay	1.9338	0.0069	507	1st Wash, The Sirens	1.8795	0.0017	7929
1st Wash, Palmer Head	1.4764	0.0874	553	1st Wash, Princess Bay	2.2098	0.0004	9142
1st Wash, Flax Bay	2.1871	0.0066	128	1st Wash, Palmer Head	1.3632	0.0832	6632
1st Wash, Barret Reef	1.4284	0.0434	8648	1st Wash, Flax Bay	2.4375	0.0007	3226
Red Rocks, Yung Pen	1.6592	0.0285	4315	1st Wash, Barret Reef	1.6524	0.0024	9451
Red Rocks, The Sirens	2.4542	0.0010	5015	1st Wash, Breaker Bay	2.2511	0.0011	4962
Red Rocks, Princess Bay	2.9050	0.0003	1861	Red Rocks, Yung Pen	1.9795	0.0036	6583
Red Rocks, Palmer Head	3.2315	0.0001	2465	Red Rocks, The Sirens	1.4603	0.0540	6662
Red Rocks, Flax Bay	3.4164	0.0002	505	Red Rocks, Princess Bay	1.7526	0.0121	8674
Red Rocks, Barret Reef	1.9758	0.0001	9451	Red Rocks, Palmer Head	1.4210	0.0708	4999
Yung Pen, The Sirens	1.4323	0.0683	740	Red Rocks, Flax Bay	2.1613	0.0062	1862
Yung Pen, Princess Bay	1.3800	0.0950	132	Red Rocks, Barret Reef	1.8203	0.0002	9144
Yung Pen, Palmer Head	1.7913	0.0180	203	Red Rocks, Breaker Bay	1.9199	0.0093	3203
Yung Pen, Flax Bay	1.7880	0.0195	48	Yung Pen, The Sirens	1.0537	0.3481	4962
Yung Pen, Barret Reef	1.7311	0.0043	7121	Yung Pen, Princess Bay	1.1532	0.2430	7863
The Sirens, Princess Bay	1.0665	0.3041	128	Yung Pen, Palmer Head	1.5133	0.0496	3219
The Sirens, Palmer Head	1.0858	0.3206	192	Yung Pen, Flax Bay	1.2765	0.1501	990
The Sirens, Flax Bay	1.2338	0.1653	32	Yung Pen, Barret Reef	2.0617	0.0001	8687
The Sirens, Barret Reef	1.8510	0.0048	6629	Yung Pen, Breaker Bay	1.1693	0.2182	1411
Princess Bay, Palmer Head	1.1646	0.2888	31	The Sirens, Princess Bay	1.1146	0.2757	7874
Princess Bay, Flax Bay	0.8745	0.8606	6	The Sirens, Palmer Head	1.3106	0.1005	3226
Princess Bay, Barret Reef	2.3697	0.0002	3255	The Sirens, Flax Bay	1.4768	0.0657	987
Palmer Head, Flax Bay	1.3015	0.1378	12	The Sirens, Barret Reef	1.8628	0.0002	8699
Palmer Head, Barret Reef	2.2160	0.0009	2164	The Sirens, Breaker Bay	1.3904	0.0847	1854
Flax Bay, Barret Reef	2.5522	0.0005	995	Princess Bay, Palmer Head	1.6518	0.0213	6624
				Princess Bay, Flax Bay	1.9520	0.0037	3263
				Princess Bay, Barret Reef	2.1049	0.0002	9413
				Princess Bay, Breaker Bay	1.8026	0.0036	5020
				Palmer Head, Flax Bay	1.3569	0.1093	504
				Palmer Head, Barret Reef	1.2213	0.1536	7912
				Palmer Head, Breaker Bay	1.2342	0.1512	990
				Flax Bay, Barret Reef	2.2465	0.0001	4953
				Flax Bay, Breaker Bay	0.9003	0.5887	256
				Barret Reef, Breaker Bay	1.7102	0.0050	6602

Appendix J, Chapter 4. Beta diversity similarities in assemblage composition for two epibenthic groups from Wellington south coast's nearshore rocky reefs Pairwise comparisons of sites and areas were generated with PERMDISP using a Jacquard presence/absence matrix (999 permutations). Bold values were considered significant at (permuted) $p < 0.05$.

Group	t-test	P (permutations)	Group	t-test	P (permutations)
Macro-algae					
AMONG-SITE COMPARISONS			AMONG-AREA COMPARISONS		
(2nd Wash,1st Wash)	0.8308	0.4790	Macro-algae		
(2nd Wash,Red Rocks)	1.2132	0.2840	(West,East)	1.8833	0.0530
(2nd Wash,Yung Pen)	0.2900	0.7690	(Central,East)	1.4350	0.1990
(2nd Wash,The Sirens)	1.5782	0.1840			
(2nd Wash,Princess Bay)	2.4440	0.0270			
(2nd Wash,Palmer Head)	0.9972	0.3570			
(2nd Wash,Flax Bay)	2.3025	0.0440			
(2nd Wash,Barret Reef)	0.7236	0.5090	Mobile macro-invertebrates		
(2nd Wash,Breaker Bay)	1.3022	0.2610	(West,Central)	1.9423	0.0690
(1st Wash,Red Rocks)	0.4413	0.6880	(West,East)	0.6112	0.5370
(1st Wash,Yung Pen)	0.6705	0.5330	(Central,East)	2.6095	0.0260
(1st Wash,The Sirens)	0.8783	0.4100			
(1st Wash,Princess Bay)	1.8651	0.0820			
(1st Wash,Palmer Head)	0.0558	0.9660			
(1st Wash,Flax Bay)	1.7200	0.1220			
(1st Wash,Barret Reef)	0.2462	0.8290			
(1st Wash,Breaker Bay)	2.5878	0.0190			
(Red Rocks,Yung Pen)	1.1392	0.2960			
(Red Rocks,The Sirens)	0.4369	0.7000			
(Red Rocks,Princess Bay)	1.3759	0.2070			
(Red Rocks,Palmer Head)	0.4849	0.6360			
(Red Rocks,Flax Bay)	1.2907	0.2240			
(Red Rocks,Barret Reef)	0.7665	0.4960			
(Red Rocks,Breaker Bay)	3.0586	0.0100			
(Yung Pen,The Sirens)	1.6097	0.1480			
(Yung Pen,Princess Bay)	2.7736	0.0140			
(Yung Pen,Palmer Head)	0.8951	0.4130			
(Yung Pen,Flax Bay)	2.4802	0.0310			
(Yung Pen,Barret Reef)	0.5329	0.6220			
(Yung Pen,Breaker Bay)	2.1412	0.0640			
(The Sirens,Princess Bay)	0.9006	0.4050			
(The Sirens,Palmer Head)	1.0426	0.3330			
(The Sirens,Flax Bay)	0.8690	0.4720			
(The Sirens,Barret Reef)	1.3012	0.2080			
(The Sirens,Breaker Bay)	3.6257	0.0020			
(Princess Bay,Palmer Head)	2.4471	0.0210			
(Princess Bay,Flax Bay)	0.1147	0.9260			
(Princess Bay,Barret Reef)	2.6462	0.0140			
(Princess Bay,Breaker Bay)	5.6696	0.0010			
(Palmer Head,Flax Bay)	2.0633	0.0600			
(Palmer Head,Barret Reef)	0.4023	0.7340			
(Palmer Head,Breaker Bay)	4.0714	0.0020			
(Flax Bay,Barret Reef)	2.2724	0.0520			
(Flax Bay,Breaker Bay)	4.3165	0.0010			
(Barret Reef,Breaker Bay)	3.2748	0.0100			

Appendix K (Chapter 5).Wet weight biomass and morphometric summaries for macro-algal and mobile macro-invertebrate species. Data were collected off Wellington, New Zealand's south coast nearshore rocky reefs in (austral) winter and summer, 2007-09.

SEASON/YEAR	WINTER ¹ 2007			SUMMER ¹ 2007-08			WINTER 2008			SUMMER 2008-09		
AREA	West	Central	East	West	Central	East	West	Central	East	West	Central	East
Macro-algae												
<i>Carpophyllum flexuosum</i>	20	Not seen	20	20	No collection	20	20	Not seen	19	20	No collection	20
<i>C. maschalocarpum</i>	20	20	20	Not seen	No collection	19	Not seen	19	20	Not seen	No collection	20
<i>Caulerpa</i> spp ³	20	20	20	20	No collection	20	20	20	n.o.	20	No collection	20
<i>Ecklonia radiata</i>	20	20	20	20	No collection	20	20	20	20	20	No collection	20
<i>Landsburgia quercifolia</i>	20	20	20	20	No collection	20	20	20	20	20	No collection	20
<i>Lessonia variegata</i>	20	21	21	20	No collection	20	20	20	20	20	No collection	20
<i>Macrocystis pyrifera</i>	Not seen	Not seen	22	Not seen	No collection	25	Not seen	Not seen	21	Not seen	No collection	25
<i>Marginariella boryana</i> ³	Not collected	Not collected	Not collected	Not collected	No collection	Not collected	14	20	20	19	No collection	20
<i>Marginariella urvilliana</i> ³	Not collected	Not collected	Not collected	Not collected	No collection	Not collected	20	Not seen	Not seen	9	No collection	20
<i>Zonaria turneriana</i> ⁴	20	20	20	-	-	-	-	-	-	-	-	-
Mobile macro-invertebrates												
<i>Evechinus chloroticus</i>	20	20	20	No collection	No collection	No collection	20	20	20	20	No collection	20
<i>Haliotis australis</i>	Not seen	20	20	No collection	No collection	No collection	Not seen	20	20	Not seen	No collection	5
<i>Haliotis iris</i>	14	20	20	No collection	No collection	No collection	Not seen	20	20	Not seen	No collection	20

¹ Austral seasons. ² *Caulerpa brownii*, *C. flexilis*. These species can grow in intermingled patches so are hard to separate without damaging filaments.

³ Collections made only in 2008 due to incorrect identification of 2007 samples. ⁴ Collections discontinued after 2007 due to highly patchy distribution.

Appendix L (Chapter 5). Plant-measuring diagrams illustrating measurement of structures for the development of wet-weight biomass prediction equations. Samples were collected from three sites along Wellington's south coast during austral winter/summer seasons in 2007-09 in 5-12m. See Table 5.1 for descriptions of features that were selected for measurement.

Appendix L, page 2. Macro-algal measuring methods.

Lessonia variegata

Tape: Total length from bottom of holdfast to longest blade.

Blue arrows: Stipe length measured at three stipes across the plant width.

Yellow arrow: Stipe girth measured midway between holdfast and top of stipes with tape held snugly but not tight around the bundle.

***Macrocystis pyrifera* (lower portion)**

Stipe lengths from top of holdfast to first blade; averaged if >1 stipe

Number of stipes

Stipe diameter measured at thickest point above the holdfast

← ***Marginariella* spp**
(*boryana*, *urvilliana*) →

Frond lengths as measured for straight and spiral forms.

Appendix M (Chapter 5). Wet and dry weights of 11 canopy and understory macro-algal and three mobile macro-invertebrate species (wet weight only) collected from Wellington' south coast. Morphometrics are also presented. Collections were made in austral winters and summers in 2007-09 from shallow (7-12m) nearshore rocky reefs. Dry weight data (macro-algae only) were collected only during winter 2007.

<i>Carpophyllum flexuosum</i> (wet weight)														
YEAR	SEASON	AREA	PLANT WET WEIGHT ¹			PLANT LENGTH			LAMINAE WEIGHT			STEM WEIGHT		
			(g)			(cm)			(g)			(g)		
			Mean	SE ²	N	Mean	SE	N	Mean	SE	N	Mean	SE	N
2007	Winter	West	256	37.6	20	122	13.7	20	NC ⁵			NC		
		East	248	45.8	19	102	10.1	19	16	2.2	19	232	44.5	19
2007_08	Summer	West	318	67.9	20	99	10.6	20	304	66.2	20	14	1.9	20
		East	204	38.2	20	79	7.5	20	189	37.1	20	15	1.5	20
2008	Winter	West	195	40.3	20	102	12.2	20	182	39.3	20	13	1.5	20
		East	295	58.5	19	105	9.5	19	270	55.5	19	24	3.4	19
2008_09	Summer	West	288	76.9	20	110	13.6	20	272	75.8	20	16	1.9	20
		East	231	54.5	20	101	12.3	20	237	54.0	20	19	5.3	20
Mean by area		West	265	29.0	80	108	6.3	80	253	36.0	60	14	1.0	60
		East	244	24.7	78	97	5.0	78	179	23.9	78	71	15.0	78

YEAR	SEASON	AREA	STEM LENGTH TO 1 ST BRANCH (cm)			STEM WIDTH ³ (cm)			MEAN BLADE LENGTH ⁴ (cm)			NUMBER OF BRANCHES ⁴		
			Mean	SE	N	Mean	SE	N	Mean	SE	N	Mean	SE	N
2007	Winter	West	7.2	0.9	20	1.5	0.1	20	NC			NC		
		East	7.7	0.6	19	1.6	0.1	19	NC			13	1.7	19
2007_08	Summer	West	7.3	1.0	20	1.5	0.1	20	NC			10	1.7	19
		East	5.3	0.7	20	1.6	0.1	20	NC			NC		
2008	Winter	West	7.2	0.9	20	1.3	0.1	20	10	0.4	20	15	2.1	20
		East	7.1	0.6	19	1.8	0.1	19	10	0.4	19	17	1.7	19
2008_09	Summer	West	6.0	0.7	20	1.3	0.1	20	11	0.5	20	9	1.3	20
		East	5.7	0.5	20	1.5	0.1	20	10	0.4	20	8	1.0	20
Mean by area		West	6.9	0.4	80	1.4	0.0	80	10	0.3	40	11	1.0	59
		East	6.4	0.3	78	1.6	0.0	78	10	0.3	39	13	1.0	58

¹ Holdfast not included in weight. ² Standard error. ³ Measured at widest point between holdfast and first branch/blade.
⁴ From 5 haphazardly selected blades. ⁵ NC = Not collected.

Appendix M (Chapter 5), page 2. Macro-algal weight and morphometric summaries, continued.

<i>Carpophyllum maschalocarpum</i> (wet weight)															
YEAR	SEASON	AREA	PLANT WET WEIGHT ¹			PLANT LENGTH			LAMINAE WEIGHT			STEM WEIGHT			
			(g)			(cm)			(g)			(g)			
			Mean	SE ²	N	Mean	SE	N	Mean	SE	N	Mean	SE	N	
2007	Winter	Central	137	17.9	20	90	4.8	20	22.1	3.2	20	67.7	5.8	20	
		East	153	33.0	20	80	7.1	20	14.2	2.0	20	66.2	7.0	20	
2007_08	Summer	Central	No collections made												
		East	120	25.6	19	84	7.7	19	12.6	1.9	19	71.8	6.7	19	
2008	Winter	Central	67	18.2	19	66	5.2	19	61.0	17.4	19	8.1	1.3	19	
		East	42	9.5	21	50	3.0	21	33.3	9.4	21	8.6	1.1	21	
2008_09	Summer	Central	No collections made												
		East	92	23.2	20	82	7.5	20	74.2	20.6	20	7.1	1.0	20	
Mean by area		Central	103	13.8	39	78	4.0	39	41.0	9.1	39	38.6	5.7	39	
		East	101	12.7	80	74	3.6	80	33.8	6.3	80	37.6	4.2	80	
	YEAR	SEASON	AREA	NUMBER OF BRANCHES ³			STEM LENGTH TO 1 ST BRANCH (cm)			STEM WIDTH (cm)					
				Mean	SE	N	Mean	SE	N	Mean	SE	N			
	2007	Winter	Central	5	0.7	20	6.9	0.8	20	1.2	0.1	20			
			East	7	0.9	20	8.8	1.2	20	1.2	0.1	20			
	2007_08	Summer	Central	No collections made											
			East	NC			8.3	1.6	19	0.9	0.1	19			
	2008	Winter	Central	NC			7.0	1.1	19	1.0	0.1	19			
			East	NC			8.7	0.9	21	1.7	0.6	21			
	2008_09	Summer	Central	No collections made											
			East	NC			7.1	0.8	20	0.9	0.1	20			
	Mean by area		Central	5	0.7	20	6.9	0.6	39	1.1	0.1	39			
			East	7	0.9	20	8.2	0.6	80	1.2	0.2	80			

¹ Holdfast not included in weight. ² Standard error. ³ Primary branches off main stem only.

¹ Holdfast not included in weight. ² Standard error. ³ Primary branches off main stem only.

Appendix M (Chapter 5), page 3. Macro-algal weight and morphometric summaries, continued.

<i>Ecklonia radiata</i> , page 1 (wet weight)													
YEAR	SEASON	AREA	PLANT LENGTH ¹ (cm)		PLANT WEIGHT (g)		HOLDFAST WEIGHT (g)		BLADE WEIGHT ³ (g)		STIPE WEIGHT ⁴ (g)		
			Mean	SE ²	Mean	SE ²	Mean	SE	Mean	SE	Mean	SE	N
2007	Winter	West	51	2.2	246	98.3	119	95.7	121	13.7	6.3	1.1	20
		Central	51	2.5	169	17.5	25	3.7	132	14.8	8.9	1.2	20
		East	55	2.4	226	24.5	54	8.5	162	18.1	10.8	1.8	20
2007_08	Summer	West	61	2.7	216	31.5	26	4.6	178	29.1	11.3	1.7	20
		Central	No collection										
		East	63	3.3	226	33.9	31	5.2	183	29.0	12.2	1.8	20
2008	Winter	West	57	2.6	118	14.4	25	4.2	83	9.8	9.7	1.8	20
		Central	52	1.7	124	12.0	24	3.8	92	10.1	7.3	0.9	20
		East	57	2.7	122	11.7	35	5.9	73	7.0	13.8	2.3	20
2008_09	Summer	West	61	3.7	242	30.6	26	4.6	209	27.1	7.5	1.4	20
		Central	No collection										
		East	57	3.8	188	39.5	27	3.8	148	34.6	13.2	3.0	20
Mean by area		West	58	1.5	206	27.3	49	24.0	148	11.9	8.7	0.8	80
		Central	51	1.5	146	11.1	24	2.6	112	9.4	8.1	0.8	40
		East	58	1.5	191	15.2	37	3.2	141	12.9	12.5	1.1	80

¹ Measured from top of holdfast to blade's apex. ² Standard error. ³ Does not include stipe. ⁴ Cut flush with holdfast and lower margin of blade.

³ Measured midway between holdfast and blade.

Appendix M (Chapter 5), page 4. Macro-algal weight and morphometric summaries, continued.

<i>Ecklonia radiata</i> , final.															
YEAR	SEASON	AREA	STIPE WIDTH ⁵ (cm)		BLADE WIDTH ⁶ (cm)		STIPE LENGTH ⁶ (cm)		STIPE + BLADE LENGTH (cm)		HOLDFAST LENGTH ⁷ (cm)		HOLDFAST WIDTH ⁸ (cm)		N
			Mean	SE	Mean	SE	Mean	SE	Mean	SE	Mean	SE	Mean	SE ²	
2007	Winter	West	1.1	0.0	5	0.2	10	0.9	33	1.5	7.3	0.5	11.1	0.5	20
		Central	1.2	0.1	6	0.2	11	1.0	37	1.9	8.6	0.5	11.5	0.5	20
		East	1.4	0.1	5	0.3	12	1.4	33	1.9	10.9	0.5	13.5	0.4	20
2007_08	Summer	West	1.2	0.1	6	0.3	14	1.4	39	3.5	11.9	0.5	10.0	0.6	20
		Central	No collection												
		East	1.1	0.1	6	0.2	16	1.4	49	2.1	12.2	0.6	8.6	0.5	20
2008	Winter	West	1.1	0.0	6	0.2	13	1.4	35	2.4	11.4	0.9	8.4	0.7	20
		Central	1.1	0.0	5	0.1	11	0.9	32	1.5	10.9	0.5	8.4	0.4	20
		East	1.2	0.1	5	0.2	14	1.6	32	1.5	12.4	0.7	10.3	0.7	20
2008_09	Summer	West	0.9	0.0	10	2.9	11	1.2	39	2.3	10.9	0.8	8.3	0.6	20
		Central	No collection												
		East	1.0	0.1	6	0.3	13	1.8	44	2.6	10.1	0.5	7.5	0.4	20
Mean by area		West	1.1	0.0	7	0.8	12	0.7	36	1.3	10.3	0.4	9.5	0.3	80
		Central	1.1	0.0	6	0.1	11	0.7	34	1.2	9.7	0.4	9.9	0.4	40
		East	1.2	0.0	6	0.1	14	0.8	39	1.3	11.4	0.3	10.0	0.4	80

⁵ Measured at thickest point between holdfast and blade. ⁶ Measured across blade mid-section. ⁷ Measured across longest axis. ⁸ Measured across shortest axis.

Appendix M (Chapter 5), page 5. Macro-algal weight and morphometric summaries, continued.

<i>Landsburgia quercifolia</i> (wet weight)																
YEAR	SEASON	AREA	PLANT WEIGHT ¹ (g)		PLANT LENGTH (cm)			LENGTH OF STEM TO 1 ST BRANCH (cm)			NUMBER OF BRANCHES ³			STEM DIAMETER ⁴ (cm)		
			Mean	SE ²	Mean	SE	N	Mean	SE	N	Mean	SE	N	Mean	SE	N
2007	Winter	West	112	25.3	59	4.5	20	2.7	0.2	20	1	0.1	20	ND ⁵		
		Central	97	16.0	52	4.8	20	2.4	0.2	9	2	0.3	20	ND		
		East	163	36.3	53	3.3	20	4.0	0.3	20	3	0.6	20	ND		
2007_08	Summer	West	144	34.3	67	5.8	20	ND			2	0.4	20	0.9	0.1	20
		Central	No collection													
		East	142	39.7	58	5.6	20	ND			1	0.1	20	0.7	0.1	20
2008	Winter	West	81	14.0	61	5.1	20	7.9	0.7	20	1	0.2	20	0.8	0.1	20
		Central	136	32.4	57	4.0	20	ND			3	0.4	20	1.0	0.1	20
		East	123	25.3	65	5.7	20	8.6	0.8	20	2	0.2	20	0.8	0.1	20
2008_09	Summer	West	160	32.4	73	5.4	20	7.1	0.5	20	1	0.2	20	0.8	0.1	20
		Central	No collection													
		East	111	18.0	60	4.9	20	8.3	0.8	20	2	0.3	20	0.7	0.1	20
Mean by area		West	124	14.0	65	2.6	80	5.9	0.4	60	2	0.1	80	0.8	0.0	60
		Central	116	18.1	55	3.1	40	2.4	0.2	9	2	0.2	40	1.0	0.1	20
		East	135	15.4	59	2.5	80	6.9	0.5	60	2	0.2	80	0.7	0.0	60

¹ Holdfast not included in weight. ² Standard error. ³ Counted directly off holdfast. ⁴ Measured midway between holdfast and first branch. ⁵ No data.

Appendix M (Chapter 5), page 6. Macro-algal weight and morphometric summaries, continued.

<i>Lessonia variegata</i> (wet weight)													
YEAR	SEASON	AREA	PLANT LENGTH (cm)		PLANT WEIGHT (g)		HOLDFAST WEIGHT (g)		TOTAL STIPES WEIGHT ² (g)		TOTAL BLADE WEIGHT (g)		N
			Mean	SE ¹	Mean	SE	Mean	SE	Mean	SE	Mean	SE	
2007	Winter	West	89	3.2	3,031	460.0	549	41.5	774	359.0	1,709	178.1	20
		Central	91	5.1	2,253	361.2	543	67.3	410	55.1	1,052	139.0	21
		East	89	3.7	2,046	247.0	538	79.9	355	73.2	1,416	212.3	21
2007_08	Summer	West	100	3.5	2,982	469.5	548	82.6	369	67.3	2,065	330.0	20
		Central	No collections										
		East	104	2.6	3,018	364.8	722	85.1	439	87.0	1,857	213.2	20
2008	Winter	West	96	5.8	2,834	512.8	842	132.0	561	112.9	1,431	278.2	20
		Central	90	4.1	2,635	579.5	540	71.0	402	76.5	789	101.4	20
		East	91	3.1	1,731	229.9	705	116.4	811	337.6	1,119	243.7	20
2008_09	Summer	West	87	5.8	2,840	511.2	678	99.2	461	105.6	1,702	317.3	20
		Central	No collections										
		East	111	5.0	4,020	730.3	993	176.2	657	154.8	2,409	426.3	20
Mean by area		West	93	2.4	2,922	240.0	654	48.4	541	98.7	1,727	140.7	80
		Central	99	2.3	2,982	271.2	542	48.2	406	46.2	923	88.1	41
		East	90	2.4	1,888	168.5	734	60.7	563	96.4	1,697	150.5	81

¹ Standard error. ² Stipes cut flush with holdfast and below blade petioles.

Appendix M (Chapter 5), page 7. Macro-algal weight and morphometric summaries, continued.

Lessonia variegata, final.																			
YEAR	SEASON	AREA	STIPE GIRTH ³ (cm)		NUMBER OF STIPES		STIPE LENGTH (cm)		HOLDFAST WIDTH ⁴ x LENGTH ⁵		NUMBER OF BLADES			BLADE LENGTH ⁸ (cm)		BLADE WIDTH ⁸ (cm)			
			Mean	SE	Mean	SE ²	Mean	SE	Mean	SE	Mean	SE	N	Mean	SE	Mean	SE ²	N	
2007	Winter	West	37	3.0	10	1.2	22	1.3	535	49.1	210	21.6	20	Data not collected					
		Central	45	4.3	12	0.9	24	1.4	552	58.2	181	18.2	21						
		East	45	2.7	10	1.1	22	1.9	622	41.5	175	25.3	21						
		West	48	5.8	13	2.8	27	0.9	606	75.6	208	30.6	20						
2007_08	Summer	Central	No collection																
		East	50	3.7	9	1.0	30	1.9	478	46.4	280	35.5	20						
		West	62	6.7	13	2.0	30	1.9	833	90.1	255	39.4	20	27.7	2.1	3.6	0.1	20	
2008	Winter	Central	49	4.6	11	1.2	34	1.9	529	69.4	142	17.7	20	No collection					
		East	42	3.0	13	2.3	31	2.1	565	48.4	168	28.6	20	31.2	1.9	3.7	0.1	20	
		West	57	5.6	16	2.2	25	1.4	660	75.4	247	40.2	20	36.5	1.5	3.9	0.1	20	
		Central	No collections																
2008_09	Summer	East	66	7.2	28	10.9	31	1.9	593	75.3	250	31.8	20	40.0	2.0	3.8	0.1	20	
		West	51	2.9	13	1.1	26	0.8	658	38.3	230	16.8	80	32.1	1.4	3.7	0.1	40	
		Central	53	2.7	11	0.8	29	1.4	538	31.4	162	12.9	41						
Mean by area		East	43	2.0	15	2.8	28	1.1	594	31.8	218	15.8	81	35.6	1.5	3.7	0.1	40	

³ Measured midway between holdfast and blades. Measuring tape held snugly but not tightly around girth. ⁴ Across shortest axis.

⁸ Averaged from five haphazardly selected blades. Different set used for width.

Appendix M (Chapter 5), page 8. Macro-algal weight and morphometric summaries, continued.

<i>Marginariella boryana</i> (wet weight)																		
SEASON, YEAR	AREA	Plant weight ¹ (g)		Stipe weight (g)		Frond weight (g)		Plant length (cm)		Stipe length to first frond		Stipe width ² (cm)		Number of fronds		Blade length (mm)		N
		Mean	SE ³	Mean	SE	Mean	SE	Mean	SE	Mean	SE	Mean	SE	Mean	SE	Mean	SE	
Winter 2008	West	198.7	26.0	8.8	1.0	189.9	25.6	85.0	3.7	12.8	1.1	1.1	0.0	15.3	1.2	48.7	3.9	20
	Central	163.7	25.4	7.2	1.0	156.5	24.5	76.2	3.9	10.3	0.7	1.4	0.1	13.0	0.8	44.8	3.6	20
	East	189.2	23.1	8.7	0.7	180.5	22.8	75.9	5.0	13.5	0.7	1.3	0.1	14.5	0.8	39.0	2.8	20
Summer 2008	West	220.2	39.2	6.3	0.7	213.9	38.9	83.5	6.4	9.7	0.7	0.9	0.1	13.0	1.1	43.9	3.3	16
	Central	<i>No collections</i>																
	East	199.3	43.2	8.5	1.1	190.8	42.7	77.3	5.3	13.4	1.4	1.0	0.1	15.1	1.1	39.3	2.1	20

¹ Holdfast not included. ² Measured at widest point above holdfast. ³ Standard error.

<i>Marginariella urvilliana</i> (wet weight)																		
SEASON, YEAR	AREA	Plant weight ¹ (g)		Stipe weight (g)		Frond weight (g)		Plant length (cm)		Stipe length to first frond		Stipe width ² (cm)		Number of fronds		Blade length (mm)		N
		Mean	SE ³	Mean	SE	Mean	SE	Mean	SE	Mean	SE	Mean	SE	Mean	SE	Mean	SE	
Winter 2008	West	291.2	53.4	7.7	1.7	283.5	52.3	75.7	6.1	12.5	1.9	1.2	0.1	14.0	1.0	35.5	3.1	20
	Central	<i>None observed</i>																
	East	625.9	69.3	19.1	1.9	606.8	67.7	90.2	4.0	15.7	1.5	1.8	0.1	19.0	0.7	35.8	2.0	20
Summer 2008	West	374.0	59.5	10.9	1.9	363.1	57.9	85.8	7.4	10.0	1.8	1.7	0.1	18.0	1.9	36.1	4.4	8
	Central	<i>No collections</i>																
	East	461.6	90.3	13.3	1.8	448.3	88.8	87.8	5.2	14.9	1.1	1.5	0.1	15.0	0.9	39.5	2.9	20

¹ Holdfast not included. ² Measured at widest point above holdfast. ³ Standard error.

Appendix M (Chapter 5), page 9. Macro-algal weight and morphometric summaries, continued.

<i>Caulerpa</i> spp. (<i>brownii</i> , <i>flexilis</i>)							
YEAR	SEASON	AREA	WET WEIGHT 0.01m ⁻² (g)			DRY WEIGHT 0.01 m ⁻² (g)	
			Mean	SE ¹	N	Mean	SE
2007	Winter	West	52.7	5.1	20	7.7	0.9
		Central	41.3	3.1	20	6.7	0.5
		East	24.1	2.1	20	3.4	0.5
2007_08	Summer	West	46.3	5.1	20	Not measured	
		Central	No collections				
		East	31.1	2.9	20		
2008	Winter	West	43.1	3.9	20		
		Central	30.7	2.4	20		
		East	None observed				
2008_09	Summer	West	49.9	3.7	20		
		Central	No collections				
		East	36.9	2.9	20		
Mean by area		West	47.9	2.3	80		
		Central	36.0	2.1	40		
		East	30.7	1.7	60		

¹ Standard error

¹ Standard error

<i>Zonaria turneriana</i>										
SEASON, YEAR	AREA	PLANT WEIGHT 0.01m ⁻² (g)			PLANT HEIGHT ² (mm)			DRY WEIGHT ³ 0.01m ⁻² (g)		
		Mean	SE ¹	N	Mean	SE	N	Mean	SE	N
Winter 2007	West	11.4	1.8	20	134	11.2	20	2.8	0.4	20
	Central	26.3	5.0	20	173	20.2	20	6.1	1.1	20
	East	18.7	4.9	20	114	9.2	20	4.7	1.0	20

¹ Standard error. ² Height of tallest plant measured from holdfast to apex.

³ Average of three dry weights.

Appendix M (Chapter 5), page 9. Macro-algal dry weights of samples collected during winter, 2007.

Carpophyllum flexuosum

AREA	Plant dry weight ¹ (g)		
	Mean	SE ²	N
West	56	8.0	20
Central	22	2.7	18
East	None observed		

Carpophyllum maschalocarpum

AREA	Plant dry weight ¹ (g)		Stem dry weight (g)		Total lamina dry weight (g)	
	Mean	SE	Mean	SE	Mean	SE
West	33	4.7	6	1.1	27.6	4.3
Central	37	4.7	7	0.9	30.3	4.4
East	44	9.5	5	0.6	38	9.5

Cystophora scalaris

AREA	Plant dry weight ¹ (g)		
	Mean	SE ²	N
West	43	5.5	20
Central	31	3.6	20
East	21	6.2	5

Ecklonia radiata

AREA	Plant dry weight (g)		Holdfast dry weight (g)		Stipe dry weight (g)		Blade dry weight (g)		
	Mean	SE	Mean	SE	Mean	SE	Mean	SE	N
West	24	2.7	4	0.5	1.2	0.9	19	2.3	19
Central	29	3.1	5	0.8	1.8	0.2	22	2.6	20
East	38	4.0	13	2.4	2.2	0.3	22	2.5	20

Landsburgia quercifolia

AREA	Plant dry weight ¹ (g)		
	Mean	SE	N
West	26	5.7	20
Central	22	3.8	20
East	36	7.5	20

Lessonia variegata

AREA	Plant dry weight ¹ (g)		Holdfast dry weight (g)		Stipes dry weight (g)		Blade dry weight (g)		
	Mean	SE	Mean	SE	Mean	SE	Mean	SE	N
West	550	51.1	159	14.0	132.3	17.1	259	51.1	17
Central	461	55.3	160	18.3	130.2	17.2	171	55.3	20
East	482	76.2	139	22.7	106	21.3	237	76.2	19

Macrocystis pyrifera

AREA	Plant dry weight ¹ (g)		Holdfast dry weight (g)		Stipes dry weight (g)		N
	Mean	SE	Mean	SE	Mean	SE	
East	171	64.4	38	15.8	133.5	49.1	22

¹ Holdfast not included in weight. ² Standard error.

Appendix N (Chapter 5). Macro-invertebrate wet weights and morphometrics. No collections were made during (austral) summer 2007-08.

<i>Evechinus chloroticus</i>							
YEAR	SEASON	AREA	WET WEIGHT ¹ (g)		TEST DIAMETER (mm)		
			Mean	SE ²	Mean	SE	N
2007	Winter	West	635	43.2	117	2.4	20
		Central	347	35.5	107	6.4	20
		East	587	69.3	113	4.7	20
2008	Winter	West	507	31.6	106	1.8	20
		Central	382	34.5	98	2.9	20
		East	569	134.6	102	3.2	20
2008_09	Summer	West	580	30.4	114	2.0	20
		East	513	28.1	108	2.3	20
Mean by area	Total	West	574	21.3	112	1.3	60
		Central	365	24.6	102	3.5	40
		East	556	50.6	108	2.1	60

¹ After draining aboral end up for 10 minutes.

² Standard error.

Appendix N (Chapter 5), page 2. Macro-invertebrate wet weights and morphometrics. No collections were made during summer 2007-08 or 2008-09.

Haliotis australis															
YEAR	SEASON	AREA	SHELL LENGTH (mm)		SHELL WIDTH (mm)		GREEN WEIGHT ² (g)		VISCERAL WEIGHT (g)		FOOT WEIGHT ³ (g)		SHELL WT (g)		
			MEAN	SE ¹	MEAN	SE	MEAN	SE	MEAN	SE	MEAN	SE	MEAN	SE	N
2007	Winter	West	None observed												
		Central	74	1.9	52	1.1	76	5.1	18	1.6	34	2.3	23	1.7	20
		East	83	0.8	57	0.6	103	3.7	23	1.7	47	1.7	32	2.5	20
2008	Winter	West	None observed												
		Central	74	1.0	51	0.6	84	3.2	14	1.2	37	1.3	21	1.5	20
		East	80	2.0	55	1.4	88	5.9	14	1.0	42	2.4	24	2.3	20
2008_09	Summer	West	None observed												
		Central	No collections made												
		East	80	2.1	54	1.6	88	5.7	17	1.5	44	3.0	20	2.7	5
Mean by area		West	None observed												
		Central	74	1.1	52	0.6	80	3.1	16	1.0	36	1.3	22	1.1	40
		East	82	1.1	56	0.8	96	3.7	19	1.2	45	1.5	28	1.8	40
¹ Standard error. ² Shell + total flesh weight. ³ Without visceral tissue.															
Haliotis iris															
2007	Winter	West	170	3.7	121	2.7	602	31.4	104	5.7	306	15.8	192	12.7	14
		Central	125	6.3	91	4.8	309	37.6	72	9.2	145	18.0	92	10.9	20
		East	145	5.1	103	3.3	485	42.0	115	11.6	200	19.0	166	21.1	19
2008	Winter	West	None observed												
		Central	116	6.4	82	4.5	295	40.6	40	4.6	160	23.3	77	10.5	20
		East	135	5.2	98	4.0	475	44.6	76	7.6	229	21.4	146	14.3	20
2008_09	Summer	West	None observed												
		Central	No collections made												
		East	137	3.5	100	3.0	462	35.6	93	8.0	216	16.6	145	12.0	20
Mean by area		West	170	3.7	121	2.7	602	31.4	104	5.7	306	15.8	192	12.7	14
		Central	121	4.5	87	3.3	302	27.3	56	5.7	153	14.6	85	7.6	40
		East	139	2.7	100	2.0	474	23.2	94	5.6	215	11.0	152	9.2	59

Appendix O (chapter 6). Voucher library for WSC benthic mapping project: Biota. Scale bars on frame are in 1cm increments. Multiple images of the same class illustrate the range of observed variation.

Group: Infauna: Burrows in sand (arrows)
Location: Lyall Bay, East
Image reference: Still 104_B

Group: Encrusting coralline alga
Location: Breaker Bay, East
Image reference: Drop 557

Group: Encrusting coralline alga
Location: Island Bay, Central
Image reference: Drop 3

Group: Geniculate coralline alga (blue) with
non-geniculate coralline alga (yellow)
Location: Lyall Bay, East
Image reference: Still 109_C

Group: Unidentified turfing alga: foliose red (yellow);
Non-geniculate coralline alga (blue)
Location: Lyall Bay, East
Image reference: Still 149_C

Group: Unidentified turfing alga:
filamentous red
Location: Island Bay, Central
Image reference: Still 55_A

Appendix O (chapter 6), page 2. Voucher library: Biota. Scale bars on frame are in 1cm increments.

Group: Understory: *Landsburgia quercifolia*
Location: Breaker Bay, East
Image reference: Drop 478

Group: Understory, mixed: 2 red algae (yellow arrows), 1 brown alga (blue arrow)
Location: Breaker Bay, East
Image reference: Still 351_B

Group: Understory: Foliose red alga
Location: Island Bay, Central
Image reference: Drop 49

Group: Understory: Unidentified green alga
Location: The Sirens, Central
Image reference: Drop 219

Group: Understory: *Caulerpa* species
Location: Lyall Bay, East
Image reference: Still 182_A

Group: Understory: *Zonaria turneriana*
Location: Palmer Head, East
Image reference: Still 586_A

Appendix O (chapter 6), page 3. Voucher library: Biota. Scale bars on frame are in 1cm increments.

Group: Understory: *Ulva lactuca* (black) with
understory foliose brown alga (blue)
Location: The Sirens, Central
Image reference: Still 217_C

Group: Understory: *Carpomitra costata*
Location: The Sirens, Central
Image reference: Still 220_C

Group: Canopy: *Carpophyllum flexuosum*
Location: Lyall Bay, East
Image reference: Still 99_B

Group: Canopy: *C. maschalocarpum*
Location: The Sirens, Central
Image reference: Still 217_C

Group: Canopy: *Ecklonia radiata*
Location: Breaker Bay, East
Image reference: Drop_557

Group: Canopy: *Ecklonia radiata*
Location: Breaker Bay, East
Image reference: Still 382_C

Appendix O (chapter 6), page 4. Voucher library: Biota. Scale bars on frame are in 1cm increments.

Group: Canopy: *Lessonia variegata*
Location: Owhiro Bay, West
Image reference: Still 214_A

Group: Canopy: *Cystophora* species (black) with geniculate coralline algae (blue)
Location: Houghton/Princess Bays, Central
Image reference: Still 316_A

Group: Canopy: *M. pyrifera* holdfast
Location: Breaker Bay, East
Image reference: Drop 557

Group: Canopy, mixed: *Carpophyllum flexuosum* (black) with *M. pyrifera* (blue)
Location: Breaker Bay, East
Image reference: Drop 478

Group: Canopy, mixed: *M. pyrifera* (blue) with *E. radiata*, *Carpophyllum* spp (black)
Location: Breaker Bay, East
Image reference: Drop 557

Group: Canopy, mixed: *E. radiata* (black) with *L. variegata* (blue)
Location: Boat Bay, West
Image reference: Still 426_A

Appendix O (chapter 6), page 5. Voucher library: Biota. Scale bars on frame are in 1cm increments.

Group: Sessile macro-invertebrates: Porifera
Location: Houghton/Princess Bays, Central
Image reference: Still 330_C

Group: Sessile macro-invertebrates: Porifera
Location: Houghton/Princess Bays, Central
Image reference: Still 329_B

Group: Sessile macro-invertebrates: Porifera
Location: Owhiro Bay, West
Image reference: Still 209_A

Group: Sessile macro-invertebrates: Porifera
Location: Boat Bay, West
Image reference: Still 411_A

Group: Sessile macro-invertebrates: Anthozoa:
Anthothoe albocincta with Bryozoa (blue)
Location: Lyall Bay, East
Image reference: Still 139_A

Group: Sessile macro-invertebrate field,
mixed: Bryozoa, Hydrozoa, Ascidia, Porifera
Location: Lyall Bay, East
Image reference: Still 176_C

Appendix P (chapter 6). Voucher library: Substratum. Scale bars on frame are in 1cm increments.

Class: Sand, fine with ripples (parallel)
Location: Palmer Head, East
Image reference: Still_591_B

Class: Sand, fine with ripples (wavy)
Location: Boat Bay, West
Image reference: Still_419_A

Class: Sand (gravel), coarse
Location: Boat Bay, West
Image reference: Still_408_A

Class: Sand, coarse with shell hash
Location: Boat Bay, West
Image reference: Still_414_A

Class: Sand, coarse, waves: shell hash on crests (arrow).
Location: Palmer Head, East
Image reference: Still_357_A

Class: Pebble field: Mixed sand/pebble
Location: Island Bay, Central
Image reference: Still_54_A

Appendix P (chapter 6), page 2. Voucher library: Substratum. Scale bars on frame are in 1cm increments.

Class: Pebble field, packed
Location: Houghton/Princess Bay, Central
Image reference: Still_344_C

Class: Cobble in mixed pebble/sand
Location: Houghton/Princess Bay, Central
Image reference: Still_5_A

Class: Cobble in mixed pebble/sand
Location: Island Bay, Central
Image reference: Still_55_A

Class: Cobble in mixed pebble/sand with sediment layer
Location: Lyall Bay, East
Image reference: Still_107_B

Class: Boulders (uncolonised) in mixed cobble, smooth texture
Location: Island Bay, Central
Image reference: Still_47_B

Class: Boulders (colonised), in mixed cobble, rough texture
Location: Moa Point, East
Image reference: Still_88_B

Appendix P (chapter 6), page 3. Voucher library: Substratum. Scale bars on frame are in 1cm increments.

Class: Boulders (colonised) in mixed cobble
low profile
Location: Breaker Bay, East
Image reference: Still 387_A

Class: Patch reef in sand, low profile
Location: Houghton/Princess Bay, Central
Image reference: Still 345_B

Class: Reef in sand, inclined, irregular surface
high profile (Horizontal view)
Location: Houghton/Princess Bay
Image reference: Still 345_A

Class: Reef/boulder with rubble: vertical,
high profile (Downward view)
Location: Lyall Bay, East
Image reference: Still 172_B

Class: Reef, mixed orientation, gutters
and depressions; **high profile**
Location: Quarry, West
Image reference: Still 452_C

Class: Reef bisected by channel with rubble,
high profile (Downward view)
Location: Houghton/Princess Bay, Central
Image reference: Still 487_B

Appendix Q (Chapter 6). Physical habitat composition by drop-camera sampling intensity. *Ten subsets of camera drops for each sampling level were randomly selected, then the frequency patterns of subsets compared to the “true” frequency pattern, as defined by the full set of camera drops, using χ^2 analyses. Images were analysed as both video and extracted still images and at two hierarchical levels of description: general (major seabed classes) and detailed (seabed subclasses. Wellington’s nearshore seabed was surveyed in 2006-08.

		Sampling intensity			
		All camera drops	Three-quarters	Half	One-quarter
number of camera drops used		605	450*	300*	150*
IMAGE ANALYSIS METHOD	HABITAT				
	SEABED MAJOR CLASSES				
Video	Sand	40.2%	39.1%	40.9%	38.8%
Video	Pebble field	30.9%	32.3%	30.8%	31.1%
Video	Cobble field	4.0%	3.9%	4.3%	3.9%
Video	Boulder field	3.1%	3.2%	3.3%	3.6%
Video	Patch reef	5.6%	5.4%	5.3%	6.2%
Video	Reef	16.2%	16.1%	15.5%	16.3%
Still frames	Sand	43.8%	43.3%	43.4%	44.9%
Still frames	Pebble field	30.7%	30.8%	30.8%	29.9%
Still frames	Cobble field	5.8%	6.0%	6.3%	5.5%
Still frames	Boulder field	4.6%	4.6%	4.0%	4.7%
Still frames	Patch reef	2.6%	2.6%	2.6%	1.7%
Still frames	Reef	12.4%	12.6%	12.9%	13.3%
	SEABED SUBCLASSES				
Video	Sand, coarse, dunes	6.8%	6.5%	6.9%	6.6%
Video	Sand, fine, rippled	33.2%	32.2%	33.8%	32.3%
Video	Pebble/sand, mixed, dunes	11.7%	11.7%	12.1%	12.2%
Video	Pebble field	19.3%	19.5%	18.9%	20.3%
Video	Cobble in pebble/sand	4.0%	3.9%	4.3%	3.9%
Video	Boulders in cobble/sand	3.1%	3.6%	3.3%	3.2%
Video	Patch reef in sand	3.6%	4.0%	3.4%	3.6%
Video	Patch reef in rubble/sand	2.5%	2.7%	2.3%	2.3%
Video	Reef/boulder in rubble	6.6%	6.3%	6.3%	6.7%
Video	Reef	9.1%	9.5%	8.8%	8.9%
Still frames	Sand, fine, rippled	33.9%	35.1%	33.6%	33.3%
Still frames	Sand, coarse, dunes	9.9%	10.1%	9.8%	10.0%
Still frames	Pebble/sand, mixed, dunes	5.8%	6.3%	5.6%	5.8%
Still frames	Pebble field	25.5%	24.0%	25.6%	25.5%
Still frames	Cobble in pebble/sand	5.6%	5.2%	6.1%	5.8%
Still frames	Boulders in cobble/sand	4.5%	4.4%	4.0%	4.5%
Still frames	Patch reef in sand	2.0%	1.3%	2.1%	2.0%
Still frames	Patch reef in rubble/sand	0.5%	0.2%	0.3%	0.4%
Still frames	Reef/boulder in rubble	3.6%	4.1%	4.0%	3.8%
Still frames	Reef	8.8%	9.2%	8.9%	8.8%