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Evaluating the effects of reserve closure on algae, invertebrate and fish assemblages at a temperate South Australian marine reserve

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Differences in the reef biota between the Point Labatt Marine Reserve and adjacent unprotected reference areas were examined following an 18-year period of protection from fishing. Quantitative measures of fish, invertebrates and algae were obtained by divers at 16 depth-stratified locations inside and outside the reserve, and the significance of differences examined using a combination of univariate (ANOVA) and multivariate (MDS) analyses. Strong depth-related differences in the composition and abundance of algae and invertebrates were observed, both inside and outside the reserve. These community differences were most pronounced in shallow near-shore waters (<10 m depth), and were largely due to variations in the abundance of a small group of species with widespread distributions. Spatial patterns in fish were not closely related to depth, and it appears that trophic linkages between fish and the underlying algal and invertebrate assemblages at Point Labatt are either weak or occur at spatial scales larger than that covered in this study. No significant reserve-related differences were detected in the abundance, diversity or community structures of algae, invertebrates and fish examined in this study. In many cases this is because the biological attributes measured were highly variable in space, and required more intensive sampling regimes to improve statistical precision. This study emphasizes the need for more robust survey designs and their timely implementation in marine conservation planning processes.

Keywords: marine protected area, monitoring, temporal variation, biodiversity, subtidal reef

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INTRODUCTION

Marine reserves are increasingly recognized as a tool to manage and conserve marine resources (Agardy, 2000; Dayton *et al.*, 2000). By offering protection from human activities, marine reserves can influence the density and size structure of marine species, particularly those organisms targeted by fishing, and can play a key role in enhancing fisheries (Mosquera *et al.*, 2000; Roberts *et al.*, 2001). The beneficial effects of marine reserves on fish and invertebrates have been demonstrated in numerous studies. These effects include increases in the biomass (Russ & Alcala, 1996; Paddack & Estes, 2000), abundance (Guidetti *et al.*, 2005; Barrett *et al.*, 2007) and size (Cole *et al.*, 1990; Rogers-Bennett *et al.*, 2002) of target organisms inside reserves. Other effects include increases in the catch rates and yields for fish outside reserves, due to the spillover of adults and larvae (Alcala & Russ, 1990; McClanahan & Mangi, 2000).

While the direct consequences of fishing prohibition in marine reserves are frequently documented, relatively few studies have considered the indirect effects of fishing on non-target species (e.g. Babcock *et al.*, 1999; Edgar & Barrett, 1999; Shears & Babcock, 2002). The indirect effects of fishing are usually more difficult to assess than direct effects (Dayton *et al.*, 1995), but broadly fall into two categories. The first is a reduction in predation pressure and/or prey availability through reductions in the abundance of the target species. The second is a change in predation pressure resulting from reductions in the size structure of the target population.

In Tasmania, the elimination of lobster and abalone fishing from marine reserves has resulted in significant increases in the average size and number of these organisms within the boundaries of reserves (Edgar & Barrett, 1999). Such changes have had marked indirect effects on algal composition and density due to shifts in grazing pressure by molluscs and echinoderms. In New Zealand, the cessation of fishing has allowed rock lobster biomass to increase, with a resultant decrease in urchin density. This in turn has led habitats to change from urchin barrens to kelp forests (Babcock *et al.*, 1999; Shears & Babcock, 2002). Similar changes have occurred on the west coast of North America, where sea otters are the top predator (Estes & Duggins, 1995; Tegner & Dayton, 2000).

These trophic cascades are only likely to occur where ecosystems are strongly affected by top-down effects, and will not occur in systems where bottom-up effects predominate. As sea urchins are a less important component of subtidal rocky reefs in southern Australian waters than they are in New Zealand or California, the effects of fishing for rock lobster or abalone cannot be simply inferred from the results of studies done elsewhere in the world. There may also be significant

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differences among regions within southern Australia (e.g. South Australia and Tasmania).

Presently there is no detailed information available on the responses of benthic communities to abalone and rock lobster fishing closures in South Australian waters. Information on the ecological effects of these high-value fisheries is particularly pressing in South Australia, as these industries may be excluded or limited at up to 19 new Marine Protected Areas, due to be established in the State's coastal waters before 2010. This study addresses the need for information on the potential ecological effects of rock lobster and abalone fishing, by examining the cumulative long-term effects of fishing prohibition on the reef biota at a remote marine reserve in the eastern Great Australian Bight.

MATERIALS AND METHODS

Study area

Point Labatt is an aquatic reserve of 278 ha, situated on South Australia's western Eyre Peninsula, approximately 50 km south of Streaky Bay (Figure 1). The reserve lies adjacent to the Point Labatt Conservation Park, and was established under the SA Fisheries Act 1982 to conserve one of Australia's largest mainland breeding colonies of Australian sea lions (*Neophoca cinerea*). Since its declaration in October 1986, public access to the reserve has been prohibited, as has fishing and the collection and removal of any marine organisms.

The marine reserve extends one nautical mile offshore from the intertidal zone and encloses a fractured granitic reef on the western side of the Calca Peninsula. This reef promontory slopes offshore to a depth of 35 m, and coarse unconsolidated sediments largely surround it. The bottom topography of the reef is complex, and fissures, 'bommies' (shallow isolated pieces of reef located a distance offshore), and overhangs occur throughout the area. This diversity of bedforms offers a variety of attachment points for sessile marine organisms including sponges, ascidians and algae, and also provides a range of habitats for motile organisms including abalone and rock lobster.

Survey design

A depth-stratified diver survey was employed to provide quantitative measures of invertebrates, fish and algae both inside and outside the reserve boundaries. The survey design involved sampling along sixteen transects (50 m length) randomly located within four depth strata (0–5 m, 5–10 m, 10–15 m, 15–20 m; Figure 1). Two transects were established within the reserve in each depth zone, while a further two transects in each depth stratum were established outside the reserve (one to the north of the reserve and the other to the south). In an effort to minimize geographical differences in community composition between sites, reference transects were located on reef bottom less than 1.5 km from the reserve boundaries. Also, as there was anecdotal evidence of fishing occurring along the reserve edge, all reserve transects were sited at least 250 m inside the reserve boundary.

Because of its exposure to southern ocean swells, favourable diving conditions at Point Labatt are rarely encountered. Accordingly, field sampling at Point Labatt was restricted to three workable days over a four-month period

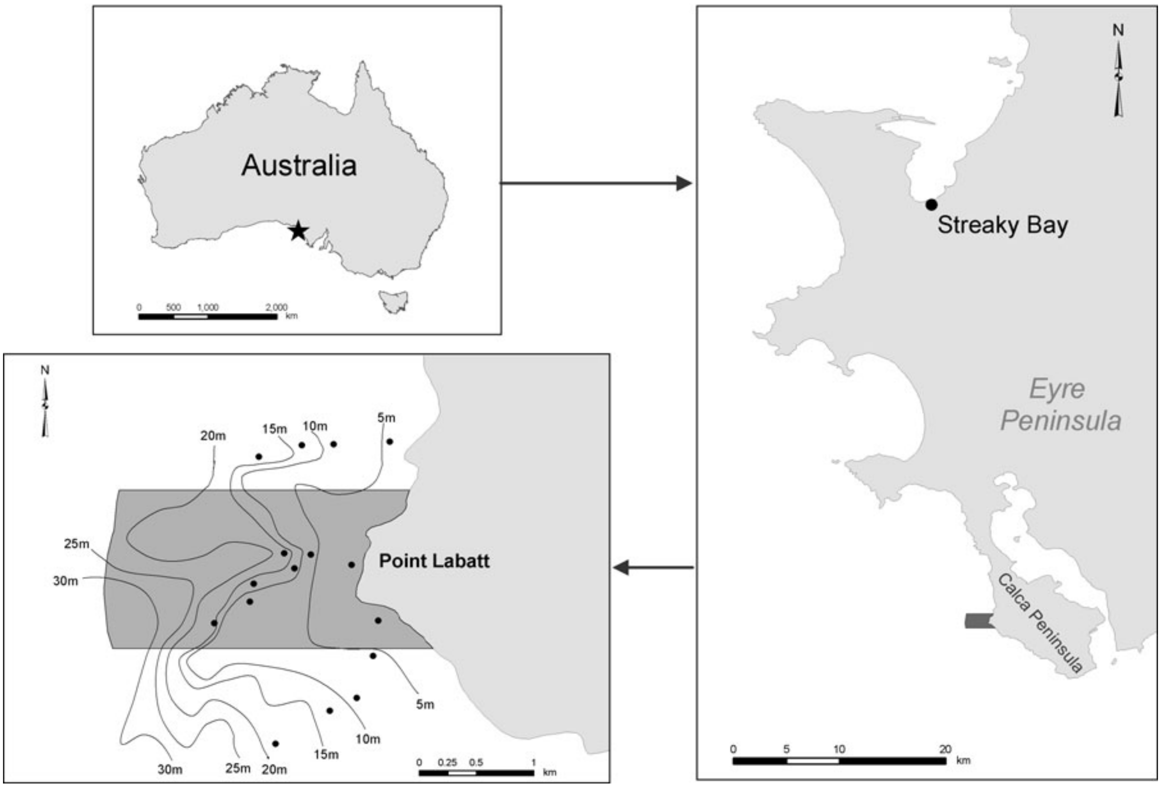

Fig. 1. Map showing the distribution of 16 study sites (solid black circles) located inside and outside the Point Labatt Marine Reserve (dark filled polygon).

(25 November 2004, 23 March 2005, 30 March 2005). On each day, transects were delineated at each site using a 50 m buoyed lead-line. This was set parallel to the prevailing wind direction using sand anchors, and the GPS coordinates for the start and endpoints recorded for future reference. Quantitative information on substrate type and the composition and densities of algae, invertebrates and fish was collected at each site by two research divers swimming one side each of the transect line.

Substrate type

Sand and cobble substrates, because of their unconsolidated nature, can often support less diverse and abundant benthic communities than adjacent areas of rocky-reef. Such differences can, in turn, affect the numbers and diversity of neighbouring reef fish (Jones, 1992). Because these interactions can potentially confound any spatial comparisons, information was collected on the composition and structure of bedforms at each transect. This was accomplished by estimating the percentage cover of three substrate classes (sand/cobble/rock) within $12^{*}1\ m^{2}$ quadrats set at random along the entire length of each transect. Estimates of rock cover were subsequently used as covariates when testing for spatial differences in biota.

Macroalgae

Estimates of algal standing-stock were determined from laboratory analysis of $12^{*}1\ m^{2}$ quadrat scrapings, taken at random from each transect swim. These algal scrapings were sorted into component taxa and identified before being weighed (wet weight, $g.m^{-2}$). Wherever possible, algae were identified to the lowest taxonomic level and voucher specimens retained for future reference in the South Australian government herbarium.

Motile invertebrates

All large motile invertebrates ($>$1 cm diameter) within a 1 m wide strip either side of the transect line were collected in catch bags by the divers and returned to the research vessel. These collections were later preserved onshore and subsequently identified, counted and weighed in the laboratory. In cases where organisms were not readily collectable due to their presence in inaccessible areas (e.g. small abalone in crevices), they were counted. Divers also counted those organisms that were particularly abundant, and where collection of all individuals in a given transect was likely to compromise bottom time.

Demersal fish

At the end of each transect, divers retraced their paths along the lead-line and recorded the identity and numbers of all fish species encountered. In this visual census divers were instructed to confine observations to a 5 m strip either side of the transect line. In cases where unfamiliar species were encountered, divers were asked to document key morphological characteristics (size, shape and colour pattern). These characteristics were subsequently used to determine the fish identity following the completion of the dive.

Data analysis

Two-way fixed factor analysis of variance (ANOVA) was used to test for differences in algal biomass both between the reserve and non-reserve sites and among different depth strata. Similar tests were also applied to examine depth and reserve-related differences in the abundance and diversity of invertebrates and fish. To avoid any issues of pseudoreplication (Hurlbert, 1987), data were aggregated at each transect prior to analysis. In all cases, homogeneity of variance was examined using Levene's test and heterogeneity removed where necessary by $\log_{10}(n + 1)$ and $1/(n + 1)$ transformations.

Depth and site-related differences in algae, invertebrate and fish community structure were also examined using Bray–Curtis dissimilarity measures (Bray & Curtis, 1957). Spatial patterns in dissimilarity were mapped using a non-metric multi-dimensional scaling (MDS) algorithm and depth and site-related differences were tested using the ANOSIM routine of Clarke & Gorley (2001). The SIMPER routine of Clarke & Gorley (2001) was subsequently used to identify those species contributing most to observed differences.

Trophic relationships between algae, invertebrates and fish were assessed by calculating rank correlation coefficients for their respective Bray–Curtis dissimilarity matrices. The null hypothesis, of no relationship between each pairwise comparison, was then tested using the RELATE permutation procedure of Clarke & Gorley (2001).

RESULTS

Macroalgae

SPECIES RICHNESS AND ABUNDANCE

A total of 73 algal species were found in the 16 transects surveyed during this study. Of these, red algae *Rhodophyta* were best represented (47 species), followed by the brown division *Phaeophyta* (21 species) and green division *Chlorophyta* (5 species). Brown algal species were the greatest contributors to standing-stock and accounted for 60% of the total algal biomass collected (73.8 kg). By contrast red and green algae accounted for just 34% and 6% of the total biomass, respectively. Much of the brown algal biomass could be attributed to two large ($>$1 m thallus length) and widely distributed kelp species. The most common of these, *Ecklonia radiata*, occurred in all but one of the transects surveyed, and individually accounted for nearly 30% of the total algal biomass. The other, *Acrocarpia paniculata*, occurred in more than 60% of the transects surveyed and accounted for a further 15% of the total algal biomass. Two species of red algae (*Phacelocarpus peperocarpus* and *Plocamium angustum*) and one species of green alga (*Caulerpa brownii*) were also relatively common at Point Labatt. These species occurred on more than half the transects surveyed, and individually accounted for more than 4% of the total algal biomass. All other species encountered (68) were rare by comparison (occurring in $<$50% of transects), and collectively contributed less than 30% to the total algal biomass.

Numbers of algal species collected in each transect varied considerably (5–26 species), however, no consistent spatial patterns were evident in plots of mean species richness (Figure 2). Measures of algal abundance (i.e. biomass) also fluctuated markedly among transects (1.8–8.6 kg/m^2), but mean distributional plots of this variable (Figure 2) were no more revealing with regards to prospective ecological gradients.

Formal two-way ANOVA tests (Table 1) failed to reveal any significant regional (inside/outside reserve) or depth-related (0–5 m, 5–10 m, 10–15 m, 15–20 m) differences in algal richness. However, the same test identified significant differences in algal biomass with depth. On closer examination (post hoc Student–Newman–Keuls test$_{\alpha=0.05}$), it was found that this result was due to disproportionately high algal biomass in shallow (0–5 m) transects only.

COMMUNITY STRUCTURE

The MDS ordination (Figure 3) maps spatial differences in the algal community structure at the 16 transects surveyed. The stress coefficient of 0.14 indicates that the ordination is not unduly distorted (Clarke, 1993), and a fair representation of the input dissimilarities in two dimensions. A pronounced depth-related gradient is readily apparent in this ordination, with shallow (0–5 m) transects plotting on the right-hand side of the ordination, deeper (5–10 m and 10–15 m) transects plotting towards the centre, and the deepest (15–20 m) transects plotting on the left-hand side of the ordination. This ordination indicates the algal community structure at Point Labatt changes progressively with increasing depth.

Analysis of similarities (ANOSIM) tests (Table 2) validate the presence of different assemblages in the four depth strata examined (Global $R = 0.541$, $P < 0.01$), and reveal that most of the observed differences in community composition occur between the 0–5 m and 5–10 m depth interval ($R > 0.5$, $P < 0.05$).

Similarity percentage (SIMPER) analyses were employed to identify those species that contributed most to similarities within and differences between the four depth strata. Biomasses of the 13 algal species contributing $\geq 5\%$ to within-strata similarity or between-strata dissimilarity for the four depth zones are given in Table 3. This table indicates that the four depth strata were characterized to some degree by relatively small subsets of species with restricted distributions. Nonetheless, it appears that much of the depth-related gradient in community structure was due to differential biomasses of a handful of widespread species. *Ecklonia radiata* and *Acrocarpia paniculata*, for example, were extremely prevalent in waters shallower than 15 m, but were rarely found in waters between 15 m and 20 m depth. Conversely, the red algal species, *Phacelocarpus peperocarpus* and *Plocamium*

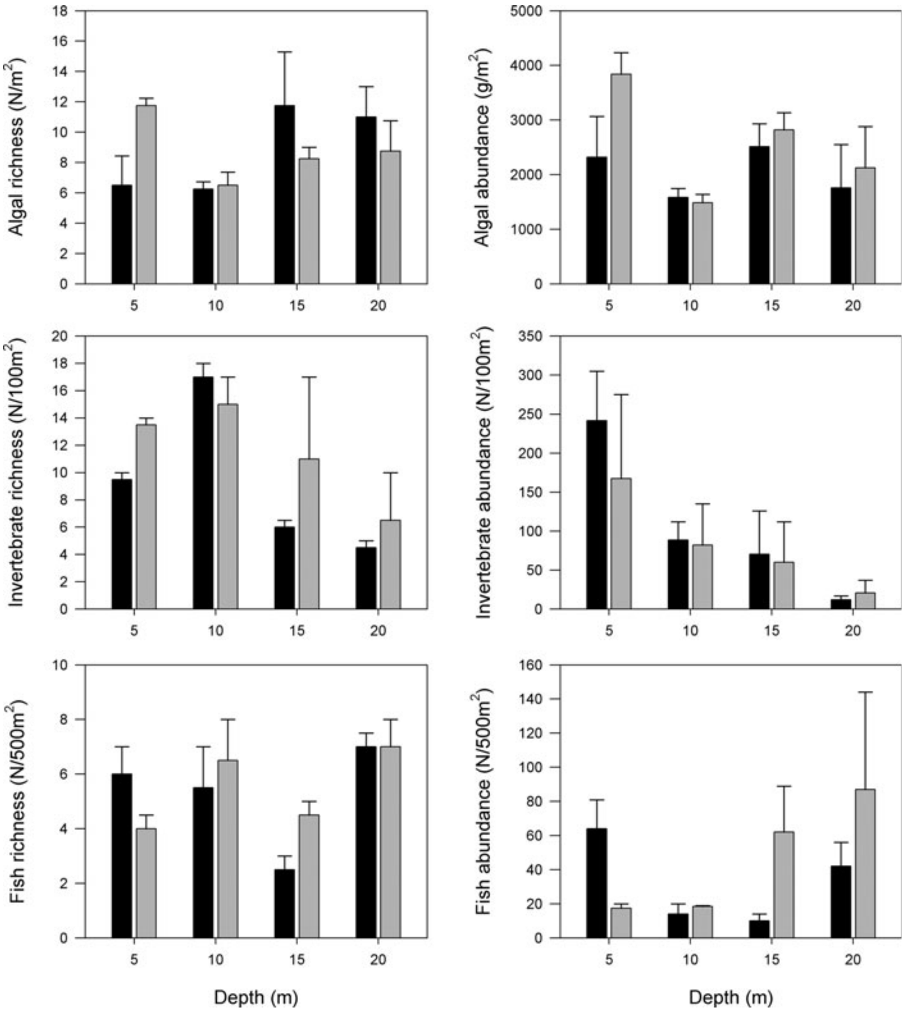

Fig. 2. Mean species richness and abundance of algae, invertebrates and fish at 16 * 50 m depth-stratified transects located inside (black bars) and outside (grey bars) the Point Labatt Marine Reserve. Error bars indicate SE of means.

Table 1. Results of two-way ANOVAs on differences in richness and abundance of algae, invertebrates and fish in four strata (depth) inside and outside the Point Labatt Marine Reserve (region). Note that the mean area of reef in each transect is included as a covariate (rock) in the ANOVA model. Power was calculated with $\alpha = 0.05$.

Taxa	Source	df	Richness				Abundance			
			MS	F	*P*	Power	MS	F	*P*	Power
Algae	Rock	1	0.079	0.006	0.940	0.051	3942398.516	3.981	0.058	0.481
	Region	1	0.058	0.004	0.948	0.050	877614.673	0.886	0.356	0.147
	Depth	3	22.390	1.642	0.207	0.372	3275632.912	3.308	0.038	0.676
	Region * depth	3	27.847	2.042	0.136	0.455	559803.587	0.565	0.643	0.148
	Error	23	13.638				990233.800			
Invertebrates	Rock	1	22.479	1.840	0.217	0.217	4.224	0.001	0.981	0.050
	Region	1	9.706	0.794	0.402	0.121	1525.828	0.214	0.658	0.069
	Depth	3	57.647	4.719	0.042	0.660	31230.375	4.370	0.049	0.484
	Region * depth	3	10.230	0.837	0.515	0.154	1277.532	0.179	0.907	0.070
	Error	7	12.217				7146.539			
Fish	Rock	1	2.814	1.761	0.226	0.210	568.098	0.469	0.515	0.092
	Region	1	0.002	0.001	0.971	0.050	415.026	0.343	0.577	0.080
	Depth	3	9.807	6.137	0.023	0.780	1747.885	1.444	0.309	0.239
	Region * depth	3	3.667	2.295	0.165	0.362	2233.468	1.845	0.227	0.298
	Error	7	1.598				1210.700			

angustum, flourished in waters between 15 m and 20 m, but were progressively less common with decreasing depth.

Differences in algal community structure between the reserve and adjacent non-reserve regions were also assessed from the species biomass ordination presented in Figure 3. In this plot, transects from the reserve and non-reserve regions intergrade considerably, and indicate that the marine reserve has no discernible influence on the community composition of algae across the study area. This absence of any significant differences in algal community structure between the reserve and non-reserve regions was formally confirmed by an ANOSIM test (Global $R = -0.02$, $P = 0.58$).

Motile invertebrates

SPECIES RICHNESS AND ABUNDANCE

A total of 1486 motile invertebrates representing 29 species from three phyla were found at the 16 transects surveyed during this study. Of the three phyla represented, echinoderms (starfish) were the most diverse (17 species, 59% of total), followed by molluscs (shellfish; 7 species, 24% of total) and crustaceans (crabs and lobsters; 5 species, 17% of total). Echinoderms were not particularly abundant, and collectively accounted for 20% of the total abundance. Molluscs, by comparison, were relatively more common (79% of the total abundance), while crustaceans were rarely encountered (1% of the total abundance).

Species richness varied among transects (3–18 species/100 m^2) but was, on average, significantly ($P < 0.05$) higher in the 5–10 m depth-zone than at all other strata (Figure 2; Table 1). In comparison, species richness was 29% lower in the 0–5 m depth stratum, and 48% and 58% lower in the 10–15 m and 15–20 m depth strata, respectively. These depth-related trends in diversity were consistent throughout the study area and did not vary significantly in relation to the placement of the marine reserve. It should, however, be noted that the statistical power associated with this test was very low (0.121; Table 1).

Invertebrate abundances varied markedly among transects (5–308 individuals/100 m^2) but declined significantly ($P < 0.05$) with increasing depth (Figure 2; Table 1). Invertebrate densities, for example, were highest on average in the shallowest 0–5 m stratum (2.04/m^2), but progressively declined through the 5–10 m ($\Delta = -58\%$), 10–15 m ($\Delta = -68\%$) and 15–20 m ($\Delta = -2\%$) depth strata. Like patterns in species richness, consistent depth-related patterns in total abundance were observed both inside and outside the reserve boundary. As a result, invertebrate abundances inside and outside the reserve were not significantly different.

Many of the observed trends in total abundance were due to the distributional patterns of three common invertebrate species. The periwinkle *Turbo undulatus*, was the most abundant species found during the study. This small (<5 cm) algal-grazing mollusc represented 43% of the total abundance, and was found almost exclusively in large aggregations in shallow nearshore waters (<5 m depth). Another mollusc, the blacklip abalone *Haliotis rubra*, accounted for a further 28% of the total abundance. This large (<20 cm) cryptic species was more widely distributed than the periwinkle, and was found in patches of variable density in most (93%) of the transects surveyed. The filter-feeding echinoderm, *Comanthus trichoptera*, accounted for a further 14% of the total fauna sampled, but like *Turbo undulatus*, this species was not widespread and was most frequently encountered in shallow waters (5–10 m depth). All other organisms (90% of species) were observed infrequently, and individually contributed less than 5% to the total abundance.

Analysis of variance tests were used to assess whether observed regional and depth-related differences in the densities of the three most common invertebrates were statistically significant. In these tests, densities of *Turbo undulatus*, *Haliotis rubra* and *Comanthus trichoptera* were not significantly ($P > 0.05$) different between the reserve and non-reserve regions. The same ANOVAs did, however, confirm that densities of two of the three most common species varied significantly with depth. As suggested, *Turbo undulatus* and *Comanthus trichoptera* were found to be significantly ($P < 0.05$) more abundant in the 0–5 m and 5–10 m depth-zones respectively, than at any other depth sampled.

Because of their relatively low abundances and generally limited distributions, it was not possible to formally test for

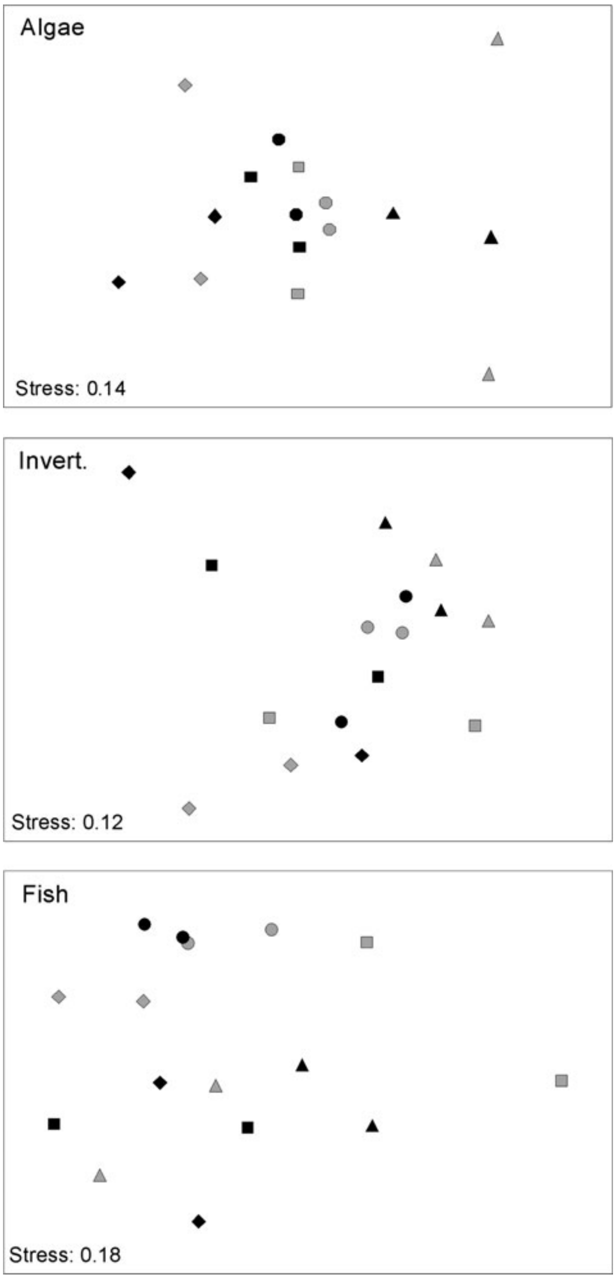

Fig. 3. Non-metric MDS plots of algae, invertebrate and fish community structure at 16 transects off Point Labatt. Depth strata are represented in the ordinations by different symbols (triangle, 0–5 m; circle, 5–10 m; square, 10–15 m; diamond, 15–20 m), and sampling regions by different shadings (black, inside the reserve; grey, outside the reserve).

regional and depth-related differences in other invertebrate species. Notably, this limitation extended to include two species of commercial importance that are actively targeted by fishermen in the area: the greenlip abalone *Haliotis laevigata*, and the southern rock lobster *Jasus edwardsii*. A total of 23 greenlip abalone were encountered during the transect survey (1600 m^2 area), 9 (39%) of which were found inside the reserve and 14 (41%) outside the reserve. Greenlip abalone were found in all depth strata surveyed, but were most frequently encountered in the deepest strata (15–20 m), where they were typically found on bare reef surfaces near sand. Rock lobsters were even rarer, and were only observed at two locations outside the reserve. One individual, estimated to be 1.5 kg, was observed in a 5–10 m transect north of the reserve boundary, while another larger specimen (∼4.0 kg) was found nearby (10–15 m transect north of the reserve boundary).

COMMUNITY STRUCTURE

Like algae, motile invertebrate community structure at Point Labatt changed progressively with depth. This trend is readily apparent in the arrangement of depth-related transect symbols in the MDS ordination presented in Figure 3. In this ordination, shallow transects (0–5 m) plot on the top right-hand side, deeper transects (5–10 m and 10–15 m) plot through the centre, while all but one of the deepest transects (15–20 m) plot towards the foot of the ordination. ANOSIM tests (Table 2) confirm the presence of different assemblages in the four depth strata examined (Global $R = 0.406$, $P < 0.01$), and reveal that most of the observed differences in community composition occur between the 0–5 m and 5–10 m depth intervals ($R > 0.5$, $P < 0.05$).

Similarity percentage (SIMPER) analyses were employed to identify those species that contributed most to similarities within and differences among the four depth strata. Abundances of the 11 invertebrate species contributing ≥5% to within-strata similarity or between-strata dissimilarity for the four depth zones are given in Table 3. The data indicate that community differences among the 0–5 m, 5–10 m, and 10–15 m depth-zones are primarily the result of differential densities in a small group of co-occurring species with extensive distributions. *Turbo undulatus*, *Haliotis rubra* and *Comanthus trichoptera*, for example, spanned the entire 0–15 m depth range, but were individually more prevalent and therefore characterized the 0–5 m, 5–10 m, and 10–15 m depth-zones respectively. By comparison, quite a different suite of invertebrate species occurred in the 15–20 m depth-zone. Neither *Turbo undulatus* nor

Table 2. Results of analysis of similarity (ANOSIM) test for differences in algal, invertebrate and fish community structure between four depth strata (0–5 m, 5–10 m, 10–15 m, 15–20 m) at Point Labatt.

Test	Groups	Algae		Invertebrate		Fish	
		R	*P*	R	*P*	R	*P*
Global	–	0.541	0.001	0.406	0.001	0.179	0.047
pairwise	0–5 m, 5–10 m	0.503	0.029	0.500	0.029	0.417	0.029
	0–5 m, 10–15 m	0.586	0.029	0.583	0.029	0.313	0.971
	0–5 m, 15–20 m	0.572	0.029	0.573	0.029	0.115	0.257
	5–10 m, 10–15 m	0.314	0.029	0.313	0.029	0.302	0.029
	5–10 m, 15–20 m	0.442	0.057	0.438	0.057	0.573	0.029
	10–15 m, 15–20 m	0.049	0.429	0.042	0.47	0.135	0.114

Table 3. Mean abundance of algae (wet weight, g per m^2) invertebrates (N per 100 m^2) and fish (N per 500 m^2) in four depth strata at Point Labatt. Species listed were identified from SIMPER analysis as contributing ≥5% to the similarity within and dissimilarity between depth groupings. Species indicative of each strata (contributing ≥10% to the total similarity within a strata) are highlighted in bold.

Group	Phylum[1]	Species	Transect depth strata (m)			
			5	10	15	20
Algae	Phaeophyta	*Ecklonia radiata*	**1194.0**	**1368.3**	**2163.1**	667.5
	Rhodophyta	*Phacelocarpus peperocarpus*	20.6	**603.7**	**1016.5**	**1620.1**
	Phaeophyta	*Acrocarpia paniculata*	**1201.8**	**485.0**	997.5	46.0
	Rhodophyta	*Plocamium angustum*	55.8	182.5	198.9	**430.8**
	Chlorophyta	*Caulerpa brownii*	216.8	107.0	282.5	63.7
	Phaeophyta	*Cystophora retorta*	580.1			1.8
	Phaeophyta	*Cystophora subfarcinata*	**540.5**			
	Rhodophyta	*Jeannerettia lobata*	256.4	51.3	8.0	1.4
	Rhodophyta	*Lenormandia smithiae*	279.8		0.5	
	Rhodophyta	*Plocamium costatum*			18.5	218.8
	Phaeophyta	*Cystophora moniliformis*	179.6	52.5		
	Phaeophyta	*Zonaria angustata*		162.1	36.0	7.3
	Rhodophyta	*Lenormandia marginata*	96.8	18.5		5.0
Invertebrates	Mollusca	*Turbo undulatus*	**149.8**	10.5	0.3	
	Mollusca	*Haliotis rubra*	27.3	20.8	**49.0**	8.3
	Echinodermata	*Comanthus trichoptera*	9.3	**35.5**	7.5	
	Mollusca	*Turbo torquatus*	7.3	4.8	2.5	0.8
	Echinodermata	*Heliocidaris erythrogramma*	3.0	3.3	0.5	
	Mollusca	*Haliotis laevigata*	0.5	1.0	1.0	**3.3**
	Echinodermata	*Nepanthia troughtoni*	0.3	**3.5**	0.3	0.8
	Mollusca	*Pleuroploca australasia*	1.0			
	Echinodermata	*Holopneustes porosissimus*			**0.8**	
	Crustacea	*Naxia spinosa*				0.8
	Echinodermata	*Meridiasta gunnii*			0.3	**0.5**
Fish	Sweep	*Scorpis aequipinnis*	**15.3**	**4.5**	**21.8**	**10.0**
	Blue-throated wrasse	*Notolabrus tetricus*	**14.8**	**5.8**	**9.3**	**14.5**
	Yellow-tail scad	*Trachurus novaezelandiae*				25.0
	Magpie perch	*Cheilodactylus nigripes*	**2.0**		1.8	2.3
	Zebra fish	*Girella zebra*	1.5			**4.3**
	Damselfish	*Parma victoriae*		**1.8**		3.0
	Blue groper	*Achoerodus gouldii*	0.5	1.0	1.0	**2.3**
	Rock cod	*Scorpaena papillosa*	2.5		0.8	
	Horseshoe leatherjacket	*Meuschenia hippocrepis*	1.0	1.0	0.8	0.3
	Moonlighter	*Tilodon sexfasciatus*	2.0	0.5		
	Herring cale	*Odax cyanomelas*	0.8		0.8	0.5

[1] Note that common fish names have been provided here to assist species recognition.

Comanthus trichoptera were observed and this deep water zone was characterized by elevated densities of greenlip abalone *Haliotis laevigata* and biscuit-stars *Meridiastra gunnii.*

Variation in invertebrate assemblage composition between the reserve and non-reserve regions was also assessed from the species abundance ordination (Figure 3). In this plot, transects from the reserve and non-reserve regions did not form distinct and separate groupings, but intergraded. This distribution suggests that the marine reserve has no discernible influence on the community composition of invertebrates across the study area. This absence of any significant differences in invertebrate community structure between the reserve and non-reserve regions was formally confirmed by an ANOSIM test (Global R = 0.056, $P = 0.19$).

Demersal fish

SPECIES RICHNESS AND ABUNDANCE

A total of 631 fish representing 17 species were observed at the 16 transects surveyed during this study. Of these, sweep *Scorpis aequipinnis* were the most common, and individually accounted for 33% of the total abundance. This schooling reef-fish was widely distributed off Point Labatt, and occurred at 81% of all transects surveyed. The next most common species, the blue-throated wrasse *Notalabrus tetricus*, accounted for a further 28% of the total abundance. This territorial reef-fish was even more widely distributed than sweep off Point Labatt, and was observed by divers at all 16 transect locations. The yellow-tailed scad *Trachurus novaezelandiae*, comprised a further 16% of the total abundance, but this fish was not widely distributed, and was only observed at one of the 16 transect sites surveyed. By comparison, most other fish species observed in Point Labatt (14) were rare. These fish species individually contributed less than 4% to the total abundance, and typically occurred in fewer than half of the transects surveyed.

Fish diversity varied little among transects (2–8 species/ 500 m^2) but was, on average, significantly ($P < 0.05$) lower in the 10–15 m depth–zone than at all other strata surveyed (Figure 2; Table 1). This statistic was primarily due to low numbers of fish species in two 10–15 m transects located within the reserve boundaries. This localized variation in richness did not, however, have a large influence on the general diversity of fish within the reserve. Specifically, we found no evidence of a significant difference in species richness between the inside and outside of the reserve, although again we stress the power for this test was low (0.05; Table 1).

Fish abundances differed considerably among transects (6–144 individuals/500 m^2) but did not vary significantly ($P > 0.05$) with depth (Figure 2; Table 1). Additionally, there was no evidence that collective fish abundances (all species combined) differed significantly across the marine reserve boundary. It appears, therefore, that total fish density across the study area is highly patchy and unaffected by either water depth or the presence of a marine reserve.

Analysis of variance tests were also used to assess the significance of regional and depth-related differences in the densities of individual fish species. Unfortunately, because most fish observed had patchy distributions, this test was restricted to just three species: sweep *Scorpis aequipinnis*, blue-throated wrasse *Notalabrus tetricus* and blue groper *Achoerodus gouldii*. In these analyses, densities of the three species did not differ significantly ($P > 0.05$) among the four depth strata surveyed, nor did they differ significantly ($P > 0.05$) across the reserve boundary.

COMMUNITY STRUCTURE

Patterns in demersal fish community structure at Point Labatt were apparently unrelated to depth. This was evident from the generally haphazard arrangement of transect symbols in the MDS ordination (Figure 3). Excepting a small cohesive grouping of circles (representing 5–10 m transects) near the top-left of this ordination, most symbols intergrade and indicate that there is considerable overlap in the types of fish species found at all depths surveyed. This homogeneity in fish species composition across all depth strata at Point Labatt was confirmed by a non-significant ANOSIM tests at the 5% level (Table 2; Global $R = 0.179$). The global statistic is nonetheless marginal because, as the MDS ordination suggests, fish assemblage structure in the 5–10 m depth strata differs from that found elsewhere.

Similarity percentage (SIMPER) analyses were employed to identify those species that contributed most to differences between the 5–10 m depth strata and all other strata combined (0–5 m + 10–15 m + 15–20 m). Results of these analyses (Table 3) show that two species (sweep and blue-throated wrasse) account for more than half of the dissimilarity between the two groups. Both of these species have wide depth distributions at Point Labatt, however, these fish are clearly less numerous in the 5–10 m depth zone than elsewhere. It is therefore apparent that this discrepancy in abundance explains most of the community differences between the two transect groupings.

Differences in fish community structure between the reserve and non-reserve regions were also examined in the species abundance ordination (Figure 3). In this plot, transects from the reserve and non-reserve regions intergraded and did not form distinct groupings. This distribution indicates that the marine reserve has no apparent influence on the community composition of fish across the study area. The absence of any significant differences in fish community structure between the reserve and non-reserve regions was formally confirmed by a non-significant ANOSIM test (Global $R = -0.056$, $P = 0.72$).

Table 4. Mantel correlation coefficients (ρ) for pairwise comparisons of algae, invertebrate and fish community structure at Point Labatt. Contrasts are based on root-transformed Bray–Curtis dissimilarity matrices that include 16 transect locations. Significance values for these correlations are provided in parentheses.

	Algae	Invertebrates	Fish
Algae	–		
Invertebrates	0.384 (0.013)	–	
Fish	0.040 (0.380)	0.032 (0.389)	–

Trophic relationships between benthos and fish

The RELATE permutation test (Table 4) confirms that there is no strong spatial relationship between fish community structure at Point Labatt and the underlaying assemblages of algae and invertebrates. In comparison, a significant correlation between algae and invertebrate community structures is identified. This shows that almost 40% of the variation in among-transect relationships for algae may be explained by among-transect relationships in invertebrates, and suggests that these two trophic groups are strongly coupled.

DISCUSSION

Spatial variability in benthos and fish

A combination of univariate and multivariate analyses has been employed in this study to assess regional and depth-related differences in benthos and fish in and around the Point Labatt Marine Reserve. Results from these two types of analysis demonstrate strong depth-related environmental gradients, including marked declines in algal biomass and invertebrate abundance and diversity with depth. However, these same analyses generally fail to identify statistically significant differences in the populations and community structures of algae, invertebrates and fish, between reserve and adjacent non-reserve areas.

Power analyses were conducted simultaneously with all ANOVAs in this study because failure to detect a real difference due to the prohibition of fishing in the reserve (type II error) was considered as serious a problem as identifying a difference when none had occurred (type I error). In most tests, it is clear that insufficient numbers of samples were collected to account for the complex patterns in abundance of most individual species and collective species groupings. We therefore caution that reserve-related difference in biota at Point Labatt quite possibly occur, but emphasize that this cannot be established with confidence given the level of sampling applied in this study. Increasing sampling replication would generally decrease the statistical error and so provide a more powerful test. However, such an action would also increase sampling costs. This information is instructive with regards to the selection of appropriate future levels of sampling effort at Point Labatt, but does not resolve inherent weaknesses in the current study.

Geological differences between the reserve and non-reserve reference sites provide another level of uncertainty. Limestone was more frequently encountered by divers outside the reserve than granite, but the extent to which recruitment and succession processes differ between these substrates is unclear. Some algal species have an increased success in attachment when rock surfaces have small grooves and pits, as opposed to smooth surfaces (Deysher & Norton, 1982). Typically, granite reefs are characterized by large slabs of bedrock with few cracks or crevices, whereas limestone reefs have many cracks, pits and fissures. In a recent study in south-western Australia, Harman *et al.* (2003) found that limestone reef supported significantly higher algal biomasses than nearby granite reef. The same study also found that algal diversity was lower on limestone than on granite. Assuming that differences in rock type influence the biota at Point Labatt, it is conceivable that these may have obscured reserve-related effects.

Visual census techniques employed to quantify fish populations in this study, represent another potential bias that may have affected the results. Visual surveys are widely used for estimating the size of fish populations (McCormick & Choat, 1987; Samoilys & Carlos, 2000), but these may be subject to a range of biases including underwater visibility, fish behaviour and diver variability (Edgar & Barrett, 1999). In this study, diver biases were unlikely to have affected the conclusions, as each diver surveyed a similar number of transects inside and outside the reserve. Similarly, visibility was not thought to have had a marked influence on the results as reserve and reference sites were sampled sequentially under similar conditions (i.e. ~10 m visibility). Fish behaviour, on the other hand, potentially represents a much larger source of uncertainty as some fish are known to modify their behaviours inside reserves and approach divers (Cole, 1994). Both blue groper *Achoerodus gouldii* and blue throated wrasse *Notolabrus tetricus*, were consistently observed following divers along transect lines within the reserve, and it is likely that densities of these species are spuriously high inside the reserve. Unfortunately the extent by which modified fish behaviours may have affected population estimates at Point Labatt is impossible to assess at the present time.

The size and extent of Australian sea lion foraging impacts on the marine biota of the study area is also unclear, but may be quite significant given the relatively high regional abundances of this mammal. Since monitoring of sea lion populations in the Point Labatt Reserve were first conducted in 1966, colony numbers have ranged between 13 and 82, and have averaged 37 annually (Department of Environment and Natural Resources, 1995). This species is reported to feed on a wide variety of prey items that are represented within the study area (including fish, lobster and molluscs; McKenzie *et al.*, 2005). It is therefore likely that sea lions exploit the local marine biota to some degree, however, prey mortalities will probably vary between seasons and years in response to a variety of factors including the population size, gender ratio, and age-structure of the colony.

Fishing impacts

Point Labatt Marine Reserve was never intended to be a fisheries management tool, however, the reserve provides some capacity to conserve and enhance stocks both locally and regionally. Movement of individuals and larvae outside the reserve may enhance local fish catches. Conversely, spillover processes may also introduce increased competition and predation rates which in-turn reduces the size-structure of the stock. Defining the ecological effects of reserve closures on fish stocks is therefore not an insignificant matter, and is further complicated by the history, intensity and spatial allocation of fishing effort. Currently the absence of any spatially explicit catch and effort data precludes a robust assessment of the effects of fishing operations around Point Labatt.

Commercial fishermen actively target three species of motile invertebrate in the waters surrounding the Point Labatt Marine Reserve. These include blacklip abalone *Haliotis rubra*, greenlip abalone *Haliotis laevigata* and rock lobster *Jasus edwardsii*. All of these fisheries are managed on annual quotas that are allocated to discrete geographical sectors along the coast.

Annual reporting and monitoring of catches occurs at a series of subunits (Marine Fishing Areas (MFAs): Mayfield *et al.*, 2004; Linnane *et al.*, 2005) within each sector. Because the MFAs for these target species at Point Labatt are large (10–100 km^2) the amount of catch or effort applied immediately adjacent to the reserve is unknown. These data are nonetheless informative with regards to regional productivity, and indicate that the area is a significant fishing ground for blacklip abalone and rock lobster but a less important ground for greenlip abalone. Between 2002 and 2004, approximately 20 tonnes of blacklip abalone, 2 tonnes of greenlip abalone, and 28 tonnes of rock lobster (representing 7%, 1% and 5% of South Australia's total commercial catch respectively) were harvested annually from reefs surrounding the reserve.

In the present survey, populations of blacklip abalone did not vary significantly between the inside and outside of the reserve, despite apparently high levels of fishing nearby. It might therefore be concluded that the cumulative effects of blacklip abalone fishing outside the reserve have been relatively minor. This reasoning is flawed however, as the population structure prior to the commencement of abalone fishing is unknown. Additionally, it is considered likely that the direct effect of fishing has been expressed primarily as a change in the size-structure of the population rather than a change in density (see Jenkins, 2004). Unfortunately, no size–frequency data were recorded during this survey to facilitate this assessment. Furthermore, little can be said about the responses of greenlip abalone and rock lobster in relation to commercial fishing pressures, as neither species were sufficiently numerous at Point Labatt to accurately assess their populations.

Recreational fishing also occurs in the waters adjoining Point Labatt Marine Reserve, although the catch and amount of fishing effort undertaken is unknown. Due to its isolation, it is considered unlikely that high levels of recreational fishing occur in the waters surrounding the reserve. Point Labatt is situated > 50 km from the nearest township Streaky Bay (population 1059) and > 20 km from the nearest boat launch site at Sceale Bay. Such large distances and associated fuel expenses are likely to be prohibitive for many recreational fishers.

Survey design

Natural resource management agencies are increasingly compelled to provide clear statements on marine reserve outcomes, and about how they will demonstrate their effectiveness over time (Carr & Raimondi, 1999). In South Australia, past marine reserve selection has been largely ad hoc (Stewart *et al.*, 2003) and the establishment of conservation and performance targets have been generally overlooked.

The identification of sensible conservation goals and the design and implementation of suitable monitoring strategies are nonetheless a critical requirement for the effective assessment of marine reserve success.

Presently there is little empirical evidence to confirm whether any observed differences in algae, invertebrates and fish between the reserve and adjacent non-reserve areas at Point Labatt are due to the prohibition of fishing within the reserve. In practice, such assessments can only be determined with a degree of certainty when appropriate spatial and temporal controls have been implemented. In the present survey, reference transects were established within 1.5 km of the reserve boundary in an effort to ensure that similar biotic assemblages would be sampled inside and outside the reserve. Results of the community analyses presented in this report attest to the fact that this has been achieved, with assemblages of algae, invertebrates and fish being largely indistinguishable across the reserve boundary. Despite this, it remains uncertain whether reference and reserve sites are adequately separated (i.e. that they are spatially uncorrelated by processes including disturbance and recruitment). In order to define the effects of fishing prohibition on the reserve, it is necessary that one or more geographically discrete reference sites be established. Ideally, these reference sites should have been established prior to the proclamation of the reserve. Moreover these reference sites should have been sampled at the same time as those within the reserve on several occasions both before and after the proclamation of the reserve. This before, after, control, impact (BACI) design (Stewart-Oaten *et al.*, 1986; Underwood, 1993) would have facilitated formal tests of all parameters for no significant change following the establishment of the reserve. However, as the design currently stands little can be said about the environmental significance of reserve-related biological differences other than that no differences were apparent.

The ambiguity of conclusions in this study emphasizes the need for more robust survey designs and their timely implementation in marine conservation planning processes. This finding has increased relevance following a commitment by the South Australian Government to develop a representative system of marine protected areas and establish 19 multiple-use marine protected areas by 2010 (Government of South Australia, 2004). The challenge now is to develop sensible objectives and assessment methodologies that are robust to the many sources of variability in natural ecosystems. Informed scientific design of practical objectives and monitoring strategies should facilitate the establishment of marine reserves, and should allow for their changing management needs as environmental conditions, external pressures and conservation requirements change over time.

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