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Short term monitoring of biotic change in Tasmanian marine reserves

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Abstract

Fishes, large invertebrates and macroalgae inside four marine reserves and at associated external reference sites off the eastern Tasmanian coast were censused between 1992 and 1993 shortly after the declaration of the reserves. Changes in several population parameters during the first year of protection in the largest Maria Island Marine Reserve were examined using two different ANOVA designs. The densities of rock lobsters and sea urchins and the mean sizes of wrasse, leatherjackets, abalone and rock lobsters all increased within the reserve relative to outside over the first year; however, only the increases in density of sea urchins and mean abalone size were statistically significant at the 5% level. The census methodology and statistical techniques nevertheless were considered sufficiently sensitive to reveal any long term change following future censuses. A doubling in population numbers of most large fishes and invertebrates, or a 10% increase in the mean size of animals, is required to indicate that significant change has occurred. Copyright © 1997 Elsevier Science B.V. All rights reserved.

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1. Introduction

On 18 September 1991 fishing was prohibited in four marine reserves (Maria Island, Tinderbox, Ninepin Point and Governor Island) off the eastern coast of Tasmania. The primary reasons for declaring these reserves were to conserve representative and unique Tasmanian marine habitats, to provide reference locations where the dynamics of marine

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communities could be observed independently of fishing effects, and to create fish propagation areas. In order to determine whether the reserves were fulfilling their intended function, a monitoring program commenced in March 1992 which involved quantitatively sampling the major reef components at a number of sites within the reserves and at external reference sites in their near vicinity. The specific aim of this program was to identify whether the creation of marine reserves in Tasmania led to an increase in abundance or changes in the size–structure of the dominant species, or whether unanticipated ecosystem effects occurred following protection. Such information is considered necessary for management, and for educating the public about benefits of marine reserves when future protected areas are proposed.

Given that marine reserves and fishery replenishment areas have been proclaimed and are widely promoted throughout Australia and the world, with a major aim of conserving or enhancing fish stocks (Davis, 1981; Kelleher and Kenchington, 1982; Roberts and Polunin, 1991; Dugan and Davis, 1993), surprisingly little quantitative information is available on the biological effects of localised fishing restrictions. While several published studies of marine reserve effects have identified increases in abundance and size–structure of particular fishes within reserves (most notably Bell, 1983; Russ and Alcala, 1989; Buxton and Smale, 1989; Bennett and Attwood, 1991), these studies may have been confounded because of intrinsic differences between sites inside reserves relative to those outside. In a review of the effects of marine reserves on reef fisheries, Roberts and Polunin (1991) could find no studies that included quantitative data on the biota of reserves prior to the imposition of fishing restrictions, and most of the existing studies compared only one site within the reserve to one site outside. This situation has been partially redressed recently, with Russ and Alcala (1996) identifying long-term increases in density and number of species of large fishes in a Philippines marine reserve; however, interpretation of that study was complicated by the likely dispersal of fishes from the reserve into external reference sites.

In this study, reef biotic data collected in 1992 and 1993 are analysed to assess the suitability of the methods used for monitoring reef communities, to compare spatial variability in important biological parameters with levels of intra- and interannual variation, and to determine if any short term (1 yr) changes are detectable within reserves following ecosystem protection. Although sampling in the study commenced nearly six months after the reserves were first proclaimed, the reserves were not signposted or policed during the first twelve months and some fishing occurred. Moreover, the benthic community was expected to respond to diminished fishing over several years given the long lifespans (> 2 yrs) of the dominant animals and the time required for new recruits to occupy the larger size-classes that had been previously exploited. Little response in the reef community attributable to the protection from fishing was therefore considered to have occurred prior to sampling.

The general approach used here for detecting changes on reefs is analogous to a replicated BACI design (Before/After Control/Impact; see Green, 1979; Stewart-Oaten et al., 1986; Underwood, 1993), which incorporates replicated sites outside ('control') and inside ('impact') reserves that are sampled 'before' and 'after' the prohibition on fishing. The important test with this design is not whether a parameter (e.g. mean abundance) is greater inside or outside marine reserves, but whether the parameter

changes with time within the reserve relative to outside. Two different ANOVA designs are compared in order to determine the most useful for identifying biological changes in Tasmanian reef communities following protection from fishing.

2. Methods

2.1. Sites examined

The locations of the four Tasmanian marine reserves and the positions of sites censused are shown in Fig. 1. Data collected at four survey times (10 March to 7 May

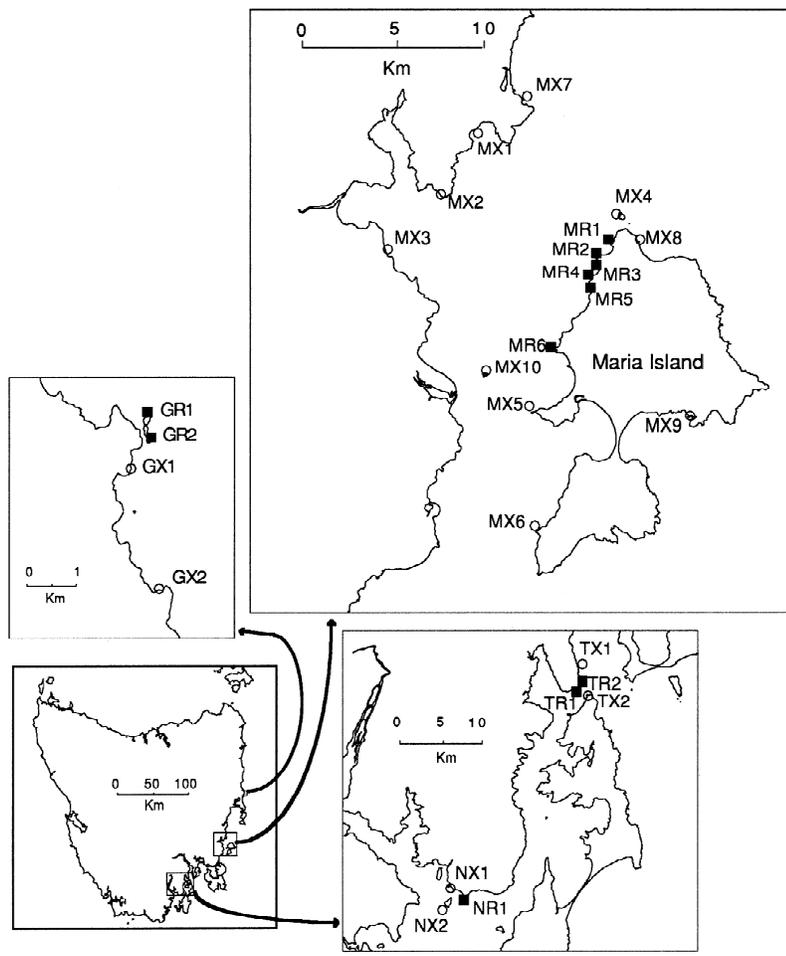


Fig. 1. Map showing distribution of study sites along the eastern Tasmanian coast. Sites with solid squares were censused within reserves, open circles indicate external reference sites.

1992, 4 July to 27 October 1992, 12 May to 8 June 1993, 30 August to 15 October 1993) were obtained from six sites protected from exploitation within the Maria Island Marine Reserve (Darlington-MR1, Magistrates Point North-MR2, Magistrates Point South-MR3, Painted Cliffs North-MR4, Painted Cliffs South-MR5, Return Point-MR6), ten external reference sites around the coast adjacent to the Maria Island Marine Reserve (Okehampton-MX1, Point Home-MX2, Spring Beach-MX3, Ile du Nord-MX4, Point Lesueur-MX5, Green Bluff-MX6, Cape Bougainville-MX7, Fossil Cliffs-MX8, Whalers Cove-MX9, Lachlan Island-MX10), two sites in the Tinderbox Marine Reserve (Tinderbox Bay-TR1, Piersons Point-TR2), two sites in close proximity to the Tinderbox Marine Reserve (Lucas Point-TX1, Dennes Point-TX2), two sites adjacent to the Ninepin Point Marine Reserve (Charlotte Cove-NX1, Huon Island-NX2) and one site in the Ninepin Point Marine Reserve (NR1). The latter reserve contained insufficient reef area for two sites to be surveyed. On all except the first of the four survey periods, an additional two sites in the Governor Island Marine Reserve (North Governor Island-GR1, Southeast Governor Island-GR2) and two sites in the vicinity (Blow Hole-GX1, Farm Point-GX2) were surveyed. The external reference sites were chosen to lie in close proximity and to encircle the reserves, and as the best match of wave exposure to the sites censused within the reserves. A related Tasmania-wide study indicated that data collected at the site spatial scale was strongly influenced by wave exposure (Edgar et al., 1997).

In order to reduce some of the spatial variability between sites, namely that related to depth, all data were obtained at 5 (± 1) m depth. This was considered the optimal depth stratum for monitoring because (i) few reefs in reserves other than Governor Island extended below 6 m so it was not possible to obtain replicated data from sufficient sites at greater depths, (ii) shallower habitats were difficult to sample because of steep slopes in some areas and wave turbulence, (iii) diving times were not limited by decompression schedules, and (iv) reefs at 5 m are subjected to heavy fishing pressure from net and rock lobster fishers and divers.

2.2. *Census methodology*

Because sampling needed to be non-destructive within the reserves and it was necessary to maximize the amount of data accumulated on a range of species within the short survey period in each season, visual census techniques were used for the study. Such techniques are widely used for estimating fish population sizes on reefs (e.g., Russell, 1977; Branden et al., 1986; McCormick and Choat, 1987), but are subject to a range of systematic biases so should not be used to estimate total densities without considerable caution (Brock, 1982; Thresher and Gunn, 1986). We use visual census data here to indicate relative rather than absolute differences between sites.

Given that Lincoln Smith (Lincoln Smith, 1988, 1989) has shown that at least two different census techniques are needed to adequately census fish, and information was also required on plants and invertebrates, three different census methods were thought necessary to obtain adequate descriptive information on reef communities. At each reef site, the abundance and size structure of large fish, the abundance of cryptic fishes and benthic invertebrates, and the percentage cover of macroalgae were each censused

separately. The densities of large fishes were estimated by laying four 50-m transect lines along the 5-m depth contour and recording on waterproof paper the number and estimated size of fish observed by a diver while swimming at an average speed of $0.2 \text{ m} \cdot \text{s}^{-1}$ along the centre of a 5-m wide swathe up one side and then down the other side of the line. A total of $4 \times 500\text{-m}^2$ transects was thus censused for large fish at each site. The distance between transects was small (0–5 m) relative to the length of transects (50 m), consequently the four transects at each site were considered subsamples which indicate variability within the site rather than as true randomly distributed replicates.

Smaller fishes and megafaunal invertebrates (large molluscs, echinoderms, crustaceans) were next counted along the transect lines used for the fish survey by recording animals within 1 m of one side of the line (a total of $4 \times 50 \text{ m}^2$ transects). The distance of 1 m was assessed using a 1-m stick carried by the diver. The maximum length of abalone and the carapace length of rock lobster were measured underwater using vernier callipers when these species were encountered. The percentage cover of macroalgal species was then assessed by placing 0.25-m^2 quadrats at 10-m intervals along the transect line, and determining the percentage cover of the various plant species by counting the number of times each species occurred directly under the 50 positions on the quadrat at which perpendicularly placed wires crossed each other (a total of 1.25 m^2 for each of the 50-m sections of transect line).

2.3. Multivariate analysis

The level of biotic similarity between reserve sites and reference sites outside reserves was examined first because there would be little point relating changes in benthic assemblages within reserves to changes at non-reserve sites if the non-reserve sites contained different communities. The data matrix showing, for each site, mean numbers of each animal species and mean percentage cover of plant species recorded during surveys was first converted to a diagonal matrix of biotic similarity between pairs of sites using the Bray-Curtis similarity index. The mean density of animals and plants in 1992 was used in this analysis, with the data $\ln(x + 1)$ transformed because multiplicative differences were thought more important than additive differences between sites (i.e., a reef with 100 animals had much greater similarity to a reef with 200 animals than a reef with 1 animal). Data for 1993 were not used in the analysis to reduce any separation of reserve sites from external sites caused by habitat protection.

The similarity matrix was agglomeratively clustered using ranked data and group-averaging, as suggested by Clarke (1993), and graphically presented using multi-dimensional scaling (MDS). Maria Island data were further examined by grouping sites within the reserve and, by using the ANOSIM procedure and the test statistic R (Clarke, 1993), determining the biotic difference between each Maria Island external site versus the grouped reserve sites.

2.4. Univariate analyses

As the primary aims of the study were to assess the value of techniques used and to provide an indication of the range of spatial and temporal variability in reef census data,

only information collected from the marine reserve of largest size and conservation importance, Maria Island, have been analysed here. Data from other areas would not be expected to show significant changes attributable to protection from fishing because of the short period of study, low replication of sites (generally $n = 2$), and low sensitivity of possible analyses.

The biological measurements examined were those considered a priori to be most interesting, namely the abundance and mean size of blue-throated wrasse *Notolabrus tetricus* (Richardson), toothbrush leatherjacket *Penicipelta vittiger* (Castelnau), southern rock lobster *Jasus edwardsii* (Hutton), sea urchin *Heliocidaris erythrogramma* (Valenciennes) and black-lip abalone *Haliotis rubra* Leach, and the number of fish species recorded during transects. *Notolabrus tetricus* was included in this analysis because it is the most abundant and widespread of the larger fish species, and *Penicipelta vittiger* because it grows rapidly with adults rarely surviving longer than two years (Barrett, 1995), so was the species most likely to show changes in population structure attributable to fishing restrictions over the short duration of the study. The three invertebrates investigated were the most heavily exploited reef species in the region. Total abundance of all fishes was not analysed because it is heavily biased by schools of small (< 10 cm) fishes, such as the plesiopid *Trachinops caudimaculatus* McCoy, that are not directly affected by fishing. Abundance and mean size data were $\ln(x + 1)$ transformed because of the importance of multiplicative effects and in order to normalise variances; species richness data were not transformed.

Data were analysed using ANOVA in two ways. Firstly, a mixed model ANOVA was used to compare data from inside and outside the Maria Island marine reserve and in different years and different seasons. Reserve (inside/outside), year (1992/1993) and season (autumn/spring) were considered fixed factors in this analysis and site a random factor nested within reserve.

The second statistical design used to detect changes within the reserve relative to outside was a two-factor orthogonal ANOVA with season and reserve fixed factors. The log difference between years (i.e. 1993–1992) for each site with transects pooled was calculated and used as the statistical parameter of interest in this analysis. Because each site rather than individual transect was repeatedly measured during the study, information on spatial variation within each site was not used.

In order to quantify the sensitivity of the two different statistical designs for detecting changes in reef communities, the power of each analysis was assessed by adding a fixed log value to 1993 data obtained at reserve sites. The size of this value was gradually increased by iteration until the F -test of the interaction term in the associated ANOVA indicated that the null hypothesis (i.e., no change has occurred within the reserve relative to outside between years) should be rejected at a probability level of 0.050. This process indicated how much change in the mean value inside the reserve is needed relative to outside before the sampling and analytical protocol will indicate that a significant change has occurred (at $\alpha = 0.05$). If the sites are resurveyed using the same experimental design when the measure has changed by this calculated value then, because of sample error ε which may equally be positive or negative, there is a 50% chance that the resultant ANOVA will produce a probability value > 0.05 and 50% chance that it will be < 0.05 . The procedure is thus equivalent to conducting a power analysis to determine

effect size and setting β , the probability of a Type II error (see Fairweather, 1991), at 0.5. This procedure is simpler than a formal power analysis, uses the distributions of the actual samples and incorporates as effect size a measure that is easily interpreted (difference between means). The change calculated by this method (i.e., reserve 1993-reference 1993-reserve 1994 + reference 1994) is expressed in terms of a change in log values, so was converted to percentage change (i.e., the percentage increase in data from 1992 levels that would be required for the ANOVA to show that a significant change has occurred within the reserve relative to outside).

3. Results

3.1. Multivariate analysis

A total of 71 fish, 24 echinoderm, 17 mollusc, 5 crustacean and 92 plant species were recorded during the study. Species recorded from a single site only were excluded from analyses, leaving 41 large fish, 17 cryptic fish, 19 echinoderm, 11 mollusc, 5 crustacean and 61 plant species that were used in the multivariate analyses. Results of the MDS using site data are shown in Fig. 2. The stress associated with this two-dimensional plot was 0.10, a value considered by Clarke (1993) to provide a good indication of relationships between sites. The four Governor Island sites separated together, as did the three Ninepin Point sites, with sites from the Maria Island and Tinderbox regions being closely related to each other but not overlapping. In general, reserve sites had greater similarity to associated external reference sites than to sites associated with other marine reserves. Hierarchical clustering of sites provided no additional information to the MDS. The four Governor Island sites separated from the rest at a Bray-Curtis similarity level of 28%, but all other sites were highly chained and grouped together at similarity levels from 48–73%.

When the six Maria Island reserve sites were grouped together and compared with individual external reference sites using the ANOSIM procedure, four of the Maria Island external sites (Lachlan Island, Cape Bougainville, Fossil Cliffs and Whaler Cove) were found to have an R -statistic = 1, and thus had a lower similarity when compared with every reserve site than were any two reserve sites when compared with each other (Clarke, 1993). The biotic assemblages at these external sites were thus distinctively different from sites within the reserve. R -statistics for the other six external sites ranged from 0.91 to 0.36.

3.2. Univariate analyses

Densities and mean sizes of animals in different seasons are shown in Figs. 3–5 and the results of the nested ANOVAs in Tables 1 and 2. Data from four of the ten external reference sites for the Maria Island reserve were excluded from ANOVAs because the number of external sites ideally should be consistent with the number of sites within reserves (see Underwood, 1981), and because these sites had assemblages quite different from those occurring within the reserve. Two sea urchins only were observed at

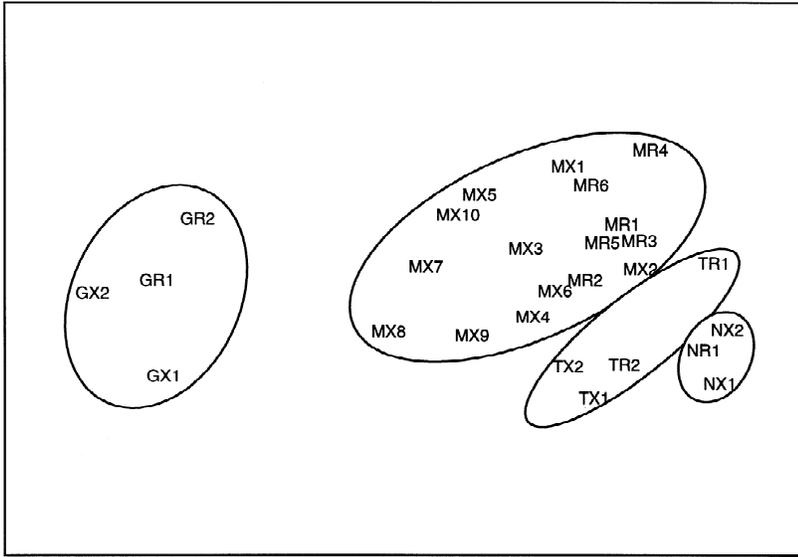


Fig. 2. Results of MDS using log abundance of fishes, invertebrates and macroalgae at different sites. Sites are as follows—Maria Island Marine Reserve: Darlington (MR1), Magistrates Point North (MR2), Magistrates Point South (MR3), Painted Cliffs North (MR4), Painted Cliffs South (MR5), Return Point (MR6), adjacent external reference sites: Okehampton (MX1), Point Home (MX2), Spring Beach (MX3), Ile du Nord (MX4), Point Lesueur (MX5), Green Bluff (MX6), Cape Bougainville (MX7), Fossil Cliffs (MX8), Whalers Cove (MX9), Lachlan Island (MX10); Tinderbox Marine Reserve: (Tinderbox Bay (TR1), Piersons Point (TR2), adjacent external reference sites: Lucas Point (TX1), Dennes Point (TX2); Ninepin Point Marine Reserve (NR1), adjacent external reference sites: Charlotte Cove (NX1), Huon Island(NX2); Governor Island Marine Reserve: North Governor Island (GR1), Southeast Governor Island (GR2), adjacent external reference sites: Blow Hole—(GX1), Farm Point—(GX2)

Okehampton during surveys, so Lachlan Island, where sea urchins were recorded in equivalent numbers to within the reserve, was included in place of Okehampton for the sea urchin analyses. ANOVAs indicated that abundances of *Penicipelta vittiger* increased significantly inside the reserve relative to outside between years; however, the details of this analysis are not provided because outlying points caused such untransformably high heteroscedasticity that the analysis was probably meaningless. Also, insufficient data on

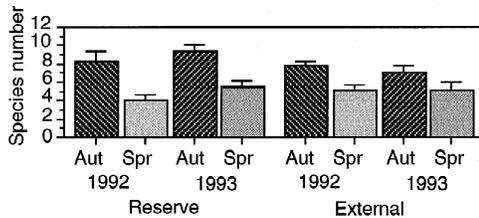


Fig. 3. Total numbers of fish species observed along 500-m² transects in autumn and spring within the Maria Island marine reserve and at external reference sites. Error bars indicate SE of means from six sites.

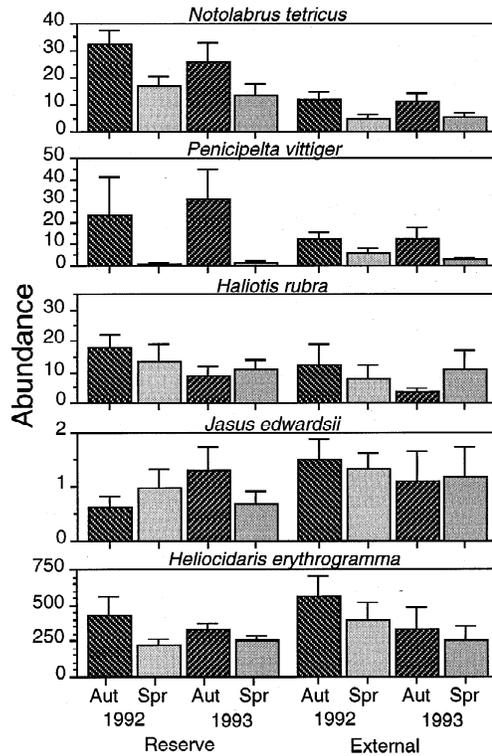


Fig. 4. Mean abundances of wrasse (*Notolabrus tetricus*) and leatherjackets (*Penicipelta vittiger*) observed along 500-m² transects, and abalone (*Haliotis rubra*), rock lobsters (*Jasus edwardsii*) and sea urchins (*Heliocidaris erythrogramma*) along 50-m² transects, in autumn and spring within the Maria Island marine reserve and at external reference sites. Error bars indicate SE of means from six sites.

the mean size of rock lobster were collected for this parameter to be analysed. No individuals of some species were sighted within a number of transects, so several missing values occurred in the analyses of mean size summarised in Table 2.

The transect method of censusing gave consistent estimates of the various parameters, with the ANOVAs sensitive at detecting seasonal and other trends in the data. Variance between sites was particularly low for the data on mean size. Much of the total variance for all parameters resided with the error term and therefore was attributable to variation between individual transects. Seasonal trends were apparent in the densities of *Notolabrus tetricus*, *Penicipelta vittiger* and *Heliocidaris erythrogramma* and the total number of species observed per transect, which all declined between autumn and spring (Table 1). A slight seasonal increase in the mean size of *P. vittiger* occurred between autumn and spring due to the recruitment of juveniles over summer.

For all analyses other than abundance of *Notolabrus tetricus*, the mean values at external reference sites at the commencement of the study were comparable to the reserve sites, so changes within the reserve can be confidently compared to changes

Table 1
Results of nested ANOVAs using log-transformed data on the number of species per transect and the abundances of wrasse (*Notolabrus tettricus*), abalone (*Haliotis rubra*), rock lobsters (*Jasus edwardsii*) and sea urchins (*Haliotidaris erythrogramma*)

Source	Hypothesis	DF	Number of species			<i>Notolabrus</i>			<i>Haliotis</i>			<i>Jasus</i>			<i>Haliotidaris</i>		
			SS	%	F	SS	%	F	SS	%	F	SS	%	F	SS	%	F
Reserve (R)	R/L	1	14.6	0.9	0.54	50.7	26.4	9.4 ^a	17.3	7.6	1.86	0.53	0.7	0.6	0.09	0	0.01
Season (S)	S/SL	1	478	30.6	106 ^c	24.9	13.0	147 ^c	0.01	0	0.01	0.35	0.5	1.14	4.49	2.0	11.55 ^b
Year (Y)	Y/YL	1	8.76	0.6	0.86	1.0	0.5	1.05	5.05	2.2	4.26	0.07	0.1	0.15	3.11	1.4	3.62
Site[Reserve](L)	R/E	10	269	17.2	8.44 ^c	53.9	28.1	17.4 ^c	93.2	40.9	17.1 ^c	8.89	11.5	2.38 ^b	95.4	43.5	16.6 ^c
RS	RS/SL	1	34.2	2.2	7.58 ^a	0.01	0	0.05	0	0	0	0.04	0.1	0.13	0.81	0.4	2.08
RY	RY/YL	1	35.9	2.3	3.52	0.66	0.3	0.69	0.05	0	0.04	1.6	2.1	3.19	4.38	2	5.09 ^a
SY	SY/SYL	1	2.76	0.2	0.24	0	0	0	5.58	2.5	5.11 ^a	0.95	1.2	3.96	1.23	0.6	0.84
SL	SL/E	10	45.1	2.9	1.41	1.7	0.9	0.55	5.04	2.2	0.93	3.07	4.0	0.82	3.89	1.8	0.68
YL	YL/E	10	102	6.5	3.20 ^b	9.6	5.0	3.09 ^b	11.9	5.2	2.18 ^a	5.01	6.5	1.34	8.61	3.9	1.50
RSY	RSY/SYL	1	0.63	0	0.06	0.01	0	0.02	0.19	0.1	0.17	0.45	0.6	1.88	0.01	0	0.01
SYL	SYL/E	10	114	7.3	3.58 ^c	4.79	2.5	1.55	10.9	4.8	2.00 ^a	2.4	3.1	0.64	14.7	6.7	2.57 ^b
Error (E)		144	459	29.4		44.5	23.2		78.4	34.5		53.7	69.7		82.6	37.7	

^a $0.01 < p < 0.05$.

^b $0.001 < p < 0.01$.

^c $p < 0.001$.

Table 2
Results of nested ANOVAs using log-transformed data on the mean size of wrasse (*Notolabrus tetricus*), leatherjackets (*Penicipelia vittiger*) and abalone (*Haliotis rubra*)

Source	<i>N. tetricus</i>			<i>P. vittiger</i>			<i>H. rubra</i>						
	Hypothesis	SS	%	DF	F	%	SS	DF	F	%	SS	DF	F
Reserve (R)	R/L	1	0.10	0.97	1.5	0	1	0.01	0	0.12	1	0.01	3.5
Season (S)	S/SL	1	0.05	1.61	0.8	1.70	1	5.66 ^a	8.9	0	1	5.66 ^a	0
Year (Y)	Y/YL	1	1.05	56.08 ^c	16.1	0.91	1	4.85 ^a	4.7	0.12	1	4.85 ^a	3.6
Site{Reserve}(L)	L/E	10	0.99	4.41 ^c	15.2	2.88	10	1.45	15.0	0.42	10	1.45	12.6
RS	RS/SL	1	0.04	1.19	0.6	0.02	1	0.06	0.1	0.02	1	0.06	0.5
RY	RY/YL	1	0.06	3.10	0.9	0.44	1	2.35	2.3	0.05	1	2.35	1.5
SY	SY/SYL	1	0.10	3.08	1.5	0.69	1	6.12 ^a	3.6	0	1	6.12 ^a	0
SL	SL/E	10	0.34	1.52	5.2	3.00	10	1.51	15.6	0.25	10	1.51	7.6
YL	YL/E	10	0.19	0.85	2.9	1.88	10	0.95	9.8	0.12	10	0.95	3.5
RSY	RSY/SYL	1	0.09	2.74	1.3	0	1	0.02	0	0	1	0.02	0
SYL	SYL/E	10	0.31	1.31	4.8	1.12	10	0.57	5.8	0.36	10	0.57	10.9
Error (E)		136	3.23	49.4		19.21	97		34.2		117		56.3

^a 0.01 < p < 0.05.

^b 0.001 < p < 0.01.

^c p < 0.001.

Table 3
 Summary of analyses of changes in marine reserves between 1992 and 1993 relative to external reference sites

	Species number	<i>Notolabrus</i> abundance	<i>Notolabrus</i> size	<i>Penicipelta</i> size	<i>Haliotis</i> abundance	<i>Haliotis</i> size	<i>Jasus</i> abundance	<i>Helicodaris</i> abundance
Change in reserve (log data)	1.73 ^a	-0.23	0.06	0.27	0.06	0.08	0.36	0.60
Actual increase (%)	28	-19	8	24	-16	6	61	49
Nested ANOVA								
Probability	0.09	0.43	0.11	0.17	0.92	0.05	0.11	0.05
Necessary increase (log data)	2.03 ^a	0.63	0.08	0.40	0.70	0.08	0.45	0.59
Necessary increase (%)	33	87	9	48	102	8	58	80
Orthogonal ANOVA								
Probability	0.08	0.31	0.11	0.15	0.95	0.17	0.22	0.01
Necessary increase (log data)	1.98 ^a	0.49	0.10	0.29	0.77	0.10	0.47	0.55
Necessary % increase	32	63	11	34	116	10	59	73

The change that has occurred between years (i.e., $\log r93 - \log c93 - \log r92 + \log c92$, where $r93$ is the geometric mean for the six sites in the reserve in 1993, $c93$ is mean for reference 1993 sites, $r92$ is mean for reserve 1992 sites, and $c92$ is mean for reference 1992 sites) is given, as are the probability that the null hypothesis that no change has occurred be rejected (as indicated by F -test), the necessary log change between years for the F -test to indicate a significant ($p = 0.05$) change, and the necessary log change between years for the F -test to indicate a significant ($p = 0.05$) change when converted from log to percentage data. The nested design, for example, will indicate that a significant ($p = 0.05$) increase in *N. ietricus* abundance has occurred if population numbers within the reserve increase by 87% from 1992 levels. The measured change within the reserve differs slightly between the two ANOVA methods because $\log(x + 1)$ transformations were used in analyses. The actual increase between years, as measured using untransformed data [i.e., $(r93 - c93 + c92)/r92 \cdot 100$], is also shown.

^a Data for the number of species per transect were not log-transformed.

Table 4
 Results of two-way orthogonal ANOVAs using log-transformed data on the difference between 1992 and 1993 in mean abundances of wrasse (*Notolabrus tetricus*), abalone (*Haliotis rubra*), rock lobsters (*Jasus edwardsii*) and sea urchins (*Heliocidaris erythrogramma*), and the mean size of wrasse, leatherjackets (*Penicipelta vittiger*) and abalone

	DF	Species		Abundance												Size					
		number		Notolabrus			Haliotis			Jasus			Heliocidaris			Notolabrus		Penicipelta		Haliotis	
		SS	F	SS	F	SS	F	SS	F	SS	F	SS	F	SS	F	SS	F	SS	F		
Reserve	1	17.90	3.32	0.37	1.10	0	0	0.49	1.64	3.05	7.27 ^a	0.04	2.82	0.25	2.26	0.04	2.04	0.03	2.04		
Season	1	1.38	0.26	0	0.01	5.74	6.60 ^a	0.18	0.58	0.40	0.96	0.06	4.30	0.39	3.48	0.06	0.03	0	0.03		
Reserve x season	1	0.32	0.06	0	0.01	0.04	0.04	0.37	1.22	0.06	0.15	0.05	3.71	0.04	0.37	0.05	0.01	0	0.01		
Error	20	107.93		6.74		17.41		6.03		8.38		0.27		2.13		0.28					

^a 0.01 < p < 0.05.

outside. However, the abundance of *N. tetricus* inside reserves was more than double the numbers at external sites, thereby complicating the interpretation of results because of the possibility that environmental conditions independent of legislative protection may affect low and high population densities differently. Two measures, the abundance of sea urchins (Table 1) and mean size of abalone (Table 2), showed significant reserve x year interactions which were consistent with the predicted increase within the reserve relative to outside. Most of this change was due to the abundance of sea urchins and the mean size of abalone decreasing outside the reserve rather than to a large increase in size within. While most other parameters also showed a relative trend for increase within the reserve, none of these trends were significant at the 5% level.

Results of the two-way orthogonal ANOVAs which incorporated data on the difference between years were similar to the results of the nested ANOVAs, with the probability levels generated from the *F*-tests analysing interannual change within the reserve relative to outside agreeing closely using the two methods (Table 3). The one exception was that the orthogonal ANOVA did not show a significant positive change in the mean size of abalone within the reserve relative to outside (Table 4). The power of both types of analysis would allow a doubling of population size between years to be detected as significant change for most abundance measures, and a 10% change in the mean size of animals would also be found to be significant (Table 3).

In the interpretation of these analyses it is important to distinguish between changes in simple means, as shown in Figs. 3–5, and changes in the log-transformed (geometric) means, as used in the ANOVA calculations. Although the observed changes in geometric means for both sea urchin abundance and abalone size were sufficient to indicate a significant effect (Table 3), neither of these measures showed increases using simple means that were of the same order. Conversely, the observed 61% increase in mean rock

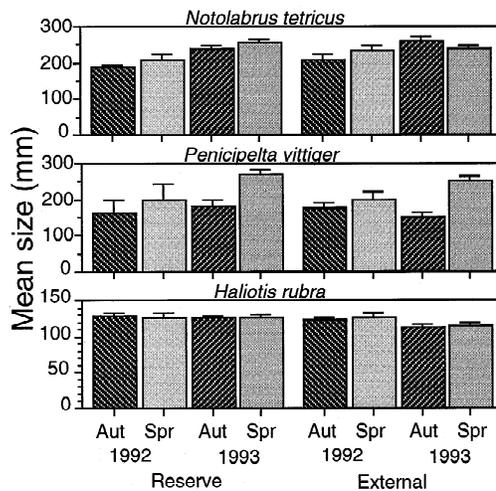


Fig. 5. Mean size of wrasse (*Notolabrus tetricus*), leatherjackets (*Penicipelta vittiger*) and abalone (*Haliotis rubra*) observed along transects in autumn and spring within the Maria Island marine reserve and at external reference sites. Error bars indicate SE of means from six sites.

lobster abundance did not translate to an increase in geometric means of 58%, the necessary level for a significant effect.

4. Discussion

Classification of sites using MDS revealed that several of the Maria Island reference sites possessed assemblages quite different from those within the reserve, despite their adjacent locations and having been selected from the map as possessing the closest environmental conditions to sites within the reserve. This discrepancy indicates the need to ensure that external reference sites are truly comparable with internal sites. It was fortunate here that more external sites were surveyed than reserve sites so that analyses into the effects of community protection did not lose statistical power when sites were removed.

On a local level, the results of the MDS were also interesting because they showed that the biota of all sites studied within the Maria Island Marine Reserve grouped closely together, and that reef assemblages within this reserve were therefore relatively homogeneous. Although the Maria Island Marine Reserve was originally proclaimed to protect representative Tasmanian east coast habitats, it apparently contains only a restricted subset of habitat types within the region. By expanding the boundaries slightly to include sites such as Fossil Cliffs (MX8), a much larger range of habitat types would be protected within the reserve.

The large proportion of total variance in each of the nested ANOVAs attributable to the error term (Tables 1 and 2) indicates a high degree of variation between individual transects within a site. If only a single 50-m transect had been surveyed at each site and used in analyses, rather than the mean value from four transects, then the total level of variation between sites would have more than doubled for most analyses and the power of the tests greatly reduced. In most cases, the number of sites surveyed would need to double or triple to retain the same power in analyses. Such changes to the experimental protocol would not be cost effective because of the substantial time spent gearing up for diving, laying new transect lines and moving the boat. Two divers were found to census fishes, invertebrates and plants along four transects in an average time of 90 min at each site, compared with a total of ≈ 100 min needed to sample one transect at a site (35 min), move the boat a distance of approximately one kilometre (30 min) and then sample another transect (35 min).

Given the different methods of calculation, the two statistical methods of analysis yielded remarkably similar results for the *F*-test of changes inside the reserve relative to outside. The main advantage of the nested ANOVA design was that it provided information on factors not included in the orthogonal analysis. Thus, the analysis of the *Notolabrus tetricus* abundance data revealed significant pre-existing differences between sites inside and outside the reserve, while the analysis of the number of species per transect data showed that different seasonal changes occurred inside relative to outside. In both cases, the results of the important reserve \times year test must be treated with a higher degree of caution because of the possibility that intrinsic differences between reserve and external sites may confound the results. The nested ANOVA also revealed

large seasonal changes in the abundance of *N. tetricus* and in the total number of species per transect. The reduction in number of large fish species per transect was presumably related to a general decline in abundance of most species following peak summer recruitment, and the reduced likelihood that an individual of a species would be encountered during a transect. Alternative explanations are that reduced species numbers were caused by an emigration of migratory species when water temperatures were coldest, or that it was a sampling artifact produced by generally lower underwater visibility during spring.

The orthogonal ANOVA nevertheless has two useful features: its simplicity of calculation and the removal of variation between sites from analysis. Because the temporal change within sites rather than the spatial variation between sites is used in calculations, the power of the two-factor method can be greater than for the nested ANOVA method when sites within a reserve are heterogeneous, for example if one site within the reserve has abundances several times greater than at most other sites.

While only two of the *F*-tests produced the predicted significant increase in abundance or mean size of animals within the reserve, large but nonsignificant trends in the predicted direction occurred for most parameters. It is therefore possible that a number of Type II errors occurred as a consequence of the short duration of the study. If the trends recorded during the 1992/1993 year were repeated over the next year then all of the measurements examined except the abundance of wrasse and abalone would show significant ($p < 0.01$) increases within the reserve relative to outside (see Table 3). Future census work using the same techniques and sites should indicate whether this assumption is correct.

The two measurements which did show significant increases within the reserve relative to outside, abundance of sea urchins and mean size of abalone, did not vary greatly between years within the reserve but declined substantially outside. The lack of a substantial increase within the reserve was expected for these species because the sea urchin fishery is a newly-developing one that is presently passing through the fish-down phase, so was at a relatively pristine state both within and outside the reserve in 1991. Most of the Tasmanian catch of sea urchins was taken from the central east coast region studied, and this catch increased from 120 tonnes in 1991 to 233 tonnes in 1992 (W. Zacharin, pers. comm.). Abalone were protected within part of the Maria Island reserve (at Darlington and Magistrates Point) prior to the imposition of total fishing restrictions, so a large increase in mean size or numbers of abalone within the reserve was also unlikely to occur.

Changes in mean size of animals within reserves appear to be much easier to detect than changes in total abundance. An increase of only 10% in the mean size of most species should be statistically significant for the data and methods of analysis described here, compared to a required doubling in abundance. Although measurements of animal size may be affected by the experience level of divers and other biases, changes in mean size are generally easier to unambiguously interpret than changes in abundance for visual census data. Significant changes in fish abundance inside reserves relative to outside may be due to changes in absolute abundance, but they may also be due to behavioural changes, with fishes attracted to divers in protected areas, artificially inflating census counts. Moreover, an increase in mean size would be expected for most

species following protection, whereas an increase in total abundance of animals may not generally occur. In the study most comparable to our Tasmanian study, Cole et al. (1990) detected no significant increase in fish abundance following marine reserve protection in northern New Zealand, although, as in Tasmania, there was a strong trend for increased numbers of rock lobsters. If intra- or interspecific competition for resources is a general feature of coastal ecosystems, then protection from human predators may result in replacement of particular suites of species without an overall density increase. Furthermore, the removal of the largest individuals that utilise a disproportionate share of resources in regions of heavy fishing may in fact free resources for large numbers of small animals, resulting in an overall density increase (see Edgar and Aoki, 1993).

In this paper, we have discussed only the value of a monitoring program for indicating any changes to reef communities that occur following legislative protection; however, it is worth noting that such census data are also potentially useful for a variety of other management purposes. A systematic survey program using methods described here can also be used to provide baseline data (i) to detect long term change due to global warming, species introductions, etc., (ii) to detect short term changes due to oil spills or other localised anthropogenic impacts, (iii) to indicate whether the local range of community types are adequately represented in the local marine reserve system, (iv) to identify any sites of special biotic significance, (v) to determine associations between plant and animal species, and (vi) to complement fishery catch data in describing the abundance and size-structure of species of economic significance (Edgar et al., 1997).

Before concluding, we would also like to stress that the use of ANOVA in this and other field studies of environmental impacts is of much less interest than the determination of means and confidence intervals (Stewart-Oaten, 1995). This is partly because reference sites are not true controls; physical conditions will always differ between each reference site and all impacted sites as a result of environmental differences, although these differences can be minimised by the careful selection of sites. More importantly, the null hypothesis tested using ANOVA, that there is no change at impacted sites relative to reference sites, is in a sense trivial. The null hypothesis will always be rejected with enough replication for measurements that are expressed as real numbers (e.g. biomass), and will usually also be rejected for integers (e.g. number of species). No two locations will possess identical densities of animals or plants or maintain constant densities over time. Significance testing using an ANOVA and field data therefore reduces to a power analysis: if a significant *F*-test is produced then sufficient replication has been used in the study; if the result is non-significant then, rather than accepting the null hypothesis, this almost invariably implies a Type II error and that further replication was needed. Our interest lies in the direction and magnitude of differences between groups of field samples, not in the fact that differences, which may well be minute, exist.

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