Research to develop and manage the sea urchin fisheries of NSW and eastern Victoria

D.G. Worthington and C.Blount

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Australia

FRDC Project No. 1999/128

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NON-TECHNICAL SUMMARY

99/128	Research to develop and manage the sea urchin fisheries of NSW and eastern Victoria

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OBJECTIVES:

(1) Develop and complete a process for stock assessment of sea urchins in NSW and eastern Victoria.

(2) Investigate techniques to enable the reliable harvesting of quality roe from coastal reefs, and determine their impact on associated species.

NON TECHNICAL SUMMARY:

OUTCOMES ACHIEVED TO DATE

This project has developed methods to assess the size of populations of sea urchins, and the quality of their roe, in NSW, eastern Victoria and Port Phillip Bay. Techniques for improving the quality of roe for marketing have also been developed and are being used by industry. The experiments involved have provided information about the potential productivity of the populations. This information has already been used in advice for determining an appropriate Total Allowable Catch for one species in NSW, and will continue to be used by fishery managers and commercial fishers. Information about the effects of reduced densities of sea urchins on other reef species is also provided, along with an overview of the market for sea urchin roe. A manual describing the sites used in the surveys is available from NSW Fisheries, so surveys can be repeated in the future. The cost of repeating the surveys, and their potential benefit to several fisheries, suggest the methods developed during this project should be useful in the ongoing management of these fisheries.

Several sea urchin fisheries exist in NSW and eastern Victoria, but have remained relatively undeveloped for many years due to high processing costs, variable roe quality and failure to develop markets. Recent expansion of the domestic market and new attempts at exporting has renewed interest in developing the existing fisheries. As sea urchin fisheries in other parts of the world have undergone rapid development and decline, estimating sustainable levels of catch for Australian fisheries prior to any rapid expansion is desirable. Further, much of the information needed to recommend sustainable catches can also be used to investigate more efficient methods of harvesting populations.

Stratified surveys were developed to estimate the density, size-structure and quality of roe for sea urchins in NSW, eastern Victoria and Port Phillip Bay. These surveys represent the first step in estimating sustainable catches in these fisheries. That is, the estimates can be combined with

estimates of the area of available habitat to calculate the probable biomass of sea urchins. In most cases, because of the limited development to date, this biomass is close to unexploited levels. Further, these estimates of population size can be combined with estimates of the likely productivity to recommend sustainable catches. In addition, they can be used to assess any changes to populations of sea urchins in response to future catches, which will provide further information about size and productivity of the populations.

The red sea urchin, *Heliocidaris tuberculata*, was found along most of the NSW coast, with the largest abundance occurring between Jervis Bay and Montague Island. Large catches of the red sea urchin were taken from this area during 2000-02. Surveys in areas open and closed to fishing suggested that the catches depleted the population by 53% between Jervis Bay and Batemans Bay and 18% between Batemans Bay and Montague Is. Large, relatively unexploited populations were observed in regions north of Jervis Bay, but only sparse populations were observed south of Montague Island. Surveys estimated a total biomass of about 1000 t of red sea urchins in NSW. The TAC Setting and Review Committee recommended a TACC of 60 t per year for red sea urchins, and suggested it should be spread among regions in proportion to biomass.

The purple sea urchin, *Centrostephanus rodgersii*, was found along most of the NSW coast, at much greater densities than the red sea urchin. Densities of the purple sea urchin were highest in areas termed Barrens, where they maintain the rock free of foliose algae. Densities were lower in areas termed Fringe, where foliose algae are abundant, and individuals with high quality roe were much more common. Surveys estimated a total biomass within 65 m of shore of about 20 000 t in Fringe and 32 000 t in Barrens. Barrens often extend much greater distances from shore, where a significant biomass would also occur.

In eastern Victoria, the purple sea urchin was present with the white sea urchin, *Heliocidaris erythrogramma*. White sea urchins were generally at much lower densities than the purple sea urchin, which occurred at similar densities to those in Fringe in NSW. For both species, only a small proportion of individuals had roe of the most preferred colour, but a large proportion of individuals still contained marketable roe. Surveys estimated a total biomass of white sea urchins of about 300 t, and purple sea urchins of about 3000 t.

In Port Phillip Bay, the white sea urchin appeared to be present wherever solid reef habitat occurred. Densities at some locations were comparable to densities of the purple sea urchin in Barrens in NSW. These locations also tended to contain individuals with roe of a poor quality. Where densities were lower (i.e. caused naturally or by fishing), a high proportion of individuals tended to contain good quality roe, unlike the same species in eastern Victoria. Surveys estimated a total biomass of white sea urchins of about 9000 t.

Reducing the density of purple sea urchins in Barrens improved the yield and colour of their roe. Although yield was improved when densities were reduced by as little as 33%, the largest increase in yield and colour (i.e. 212% and 133%) occurred when 66% of sea urchins were removed. Although yield was not increased to levels in Fringe over the time period of the experiments, it was improved to a level appropriate for market. Changes to the demography of purple sea urchin were also observed in response to the reductions in density. Although fecundity of the population was reduced, reductions in density increased the recruitment of juveniles and the rate of growth of the remaining sea urchins. These compensatory relationships are likely to increase the resilience of the population.

Transplanting purple sea urchins from Barrens to Fringe also improved the yield and colour of their roe. Transplanting was able to improve yield and colour to levels similar to those in Fringe. Following transplant, roe changed rapidly (i.e. within 3 months), particularly during spring and

summer, and when sea urchins were transplanted at densities equal to or less than natural densities in Fringe.

Reducing the density and transplanting purple sea urchins caused changes to the assemblage of benthic algae. Lower densities of purple sea urchin are associated with reductions in the cover of crustose coralline algae and increases in filamentous and foliose algae. Increases in density have the opposite effect. Surveys of macro-invertebrates living beneath purple sea urchins across most of their distribution in NSW, found over 100 taxa but most species appear to have some flexibility in their habitat requirements, as they were also abundant under rocks. An experiment done by The Centre for Research on the Ecological Impacts of Coastal Cities at the University of Sydney found that reducing the density of purple sea urchins can also reduce the local density or diversity of macro-invertebrates. Whilst at some sites, some species of macro-invertebrate do not remain when sea urchins are removed, it is unclear to what extent they simply move to adjacent habitats or are exposed to increased mortality. Despite the limitations of this latter experiment, the results suggest that managers of the sea urchin fishery need to consider responses to minimise the ecological effects of fishing on macro-invertebrate assemblages.

KEYWORDS:
Sea urchin, stock assessment, biomass, enhancement, roe quality, south-east Australia

1. INTRODUCTION

1.1. Background

There have been several attempts to develop the fishery for sea urchins in NSW and Victoria over the last 30 years. Difficulties with the reliable harvesting of good quality roe, and as a consequence the costs of processing, have prevented the development of the fishery. During this time three species have been commercially exploited: the purple (*Centrostephanus rodgersii*), red (*Heliocidaris tuberculata*) and white or green (*Heliocidaris erythrogramma*) sea urchin. The white sea urchin has also been commercially fished in Tasmania, Victoria and South Australia, but commercial quantities of purple and red urchins are restricted to NSW and eastern Victoria. There are currently 37 endorsements in the Sea Urchin and Turban Shell fishery in NSW and 24 in eastern Victoria, and in each state there has never been any large-scale survey of the stocks. With the decline in several other sea urchin fisheries around the world in recent years, there is again interest in the development of these fisheries, and landings of all species are increasing. Substantial capital investments in processing plants have been made in both NSW and Victoria, with the aim of supplying both local and international markets with sea urchin roe.

Because of the limited development of fisheries for sea urchins in NSW and Victoria, there existed an ideal opportunity to assess stocks of sea urchins prior to any major depletion by fishing. Sampling techniques were already developed for *Centrostephanus* in barren habitats (Andrew *et al.* 1998), and were easily transferred to the other urchin species and major habitat types, where commercial fishing was expected to be concentrated. Such surveys may be particularly important in the future considering the evidence from other urchin fisheries, where large unexploited stocks have been rapidly depleted with only low rates of recovery from the recruitment of juveniles (e.g. Andrew *et al.* 2002, Sanderson *et al.* 1996). Information from the independent surveys could also be combined with other information on the population (e.g. age and growth) and fishery (e.g. catch rates of different quality roe) to enable a stock assessment process for the NSW sea urchin fishery, similar to that already devised for the abalone fishery (Worthington *et al.* 1998).

The purple urchin is the dominant herbivore on coastal rocky reefs in NSW and eastern Victoria. This urchin is able to maintain areas free of macro-algae by grazing their propagules, and these areas are referred to as Barrens. Few abalone inhabit these areas, and there are concerns that past increases in the distribution of sea urchins, particularly to more southern areas, have contributed to declines in the NSW abalone fishery. A recent study of the area of Barrens on the NSW coast found no evidence of expansion over the last four years (Andrew *et al.* 1998). Removal of urchins from these areas is also likely to have a direct positive effect on abalone populations (Andrew *et al.* 1998). As a result, if the development of a fishery for sea urchins could be controlled, it could also lead to significant benefits to the abalone fishery, particularly in NSW, but also eastern Victoria.

Preliminary information from NSW suggested a large proportion of the sea urchin population, particularly in Barrens, does not contain good quality roe (Andrew *et al.* 1998). Unless good quality roe can be reliably collected, the costs of processing sea urchins may restrict development of the fishery. Two main techniques have been used in other sea urchin fisheries to improve the quality of roe harvested (Sanderson *et al.* 1996). First, by exposing young sea urchins to a higher quality of food in the wild, they can be encouraged to grow more quickly and produce roe that is of a better quality. This has been achieved by both removals of older urchins from dense populations, and the transplantation of young urchins to areas with few older urchins. Both these techniques have been successful in producing large increases in the yield and quality of sea urchin roe in other fisheries. If these techniques could be adapted for use with sea urchins in NSW, significant

improvements in yield and value would be possible. Secondly, in other fisheries yield and quality of urchin roe has been improved by holding sea urchins in sea water tanks for short periods (Tegner 1989). Although preliminary research suggests this is possible for sea urchins in NSW and Victoria, this form of enhancement was not the focus of this project. However, it was the subject of a major research project in South Australia, also funded by FRDC, entitled 'Post-harvest enhancement of sea urchin roe for the Japanese market' (FRDC Project No. 99/319).

1.2. Need

With the decline in several other sea urchin fisheries around the world, there now exists a good opportunity to develop a large and valuable fishery for purple and red sea urchins in NSW. In addition, there is also interest in the further development of the purple urchin and white urchin fishery near Mallacoota in eastern Victoria, and the white urchin fishery in Port Phillip Bay. This interest is evidenced in both NSW and Victoria by substantial capital investment in factories to process sea urchins and their roe. If these urchin fisheries could be further developed within an appropriate management framework, it could also lead to significant benefits for the abalone fishery, particularly in NSW and eastern Victoria, because of the interaction between abalone and purple sea urchins.

Because of the limited development of this fishery, an ideal opportunity exists to assess stocks of sea urchins prior to any major depletion by fishing. Sampling techniques have already been developed for purple sea urchins in barren habitats, and could easily be transferred to the other urchin species and major habitat types, where commercial fishing will initially be concentrated. Such surveys may be particularly important considering the evidence from other urchin fisheries, where large virgin stocks have been rapidly depleted with only low rates of recovery from the recruitment of juveniles.

Preliminary information from NSW suggested a large proportion of the sea urchin population, particularly in Barrens, does not contain good quality roe. Unless good quality roe can be reliably collected, the costs of processing sea urchins may restrict development of the fishery. Two main techniques have been used in other sea urchin fisheries to improve the quality of roe harvested. If these techniques could be adapted for use with sea urchins in NSW and Victoria, significant improvements in yield and value would be possible.

1.3. Objectives

(1) Develop and complete a process for stock assessment of sea urchins in NSW and eastern Victoria.

(2) Investigate techniques to enable the reliable harvesting of quality roe from coastal reefs, and determine their impact on associated species.

2. DEVELOPMENT OF METHODS TO ASSESS STOCKS OF SEA URCHIN IN NSW AND EASTERN VICTORIA

2.1. General introduction

Although fisheries for sea urchins have existed in Australia for many years, most remain relatively undeveloped in comparison to those found elsewhere in the world (see Andrew *et al.* 2002 for review). This is due to many factors, including high processing costs, variable roe quality and a failure to establish regular export markets for Australian sea urchin roe. However, recent expansion of the domestic market in Australia, and new attempts at exporting, has renewed interest in developing the existing fisheries. These include the Sea Urchin and Turban Shell restricted fishery that permits harvest of several species of sea urchin throughout NSW, and, in Victoria, several independent sea urchin fisheries, including those in eastern Victoria and Port Phillip Bay.

Surveys of abundance and biomass are completed independent of many fisheries (e.g. Worthington *et al.* 2001). Well designed surveys can provide accurate and precise information on the absolute abundance or biomass of a population and, with observations of change in response to known catches, can also help estimate the productivity of a population (e.g. Francis 1993). Such information can then be used to estimate the effects of past catches and the likely effects of any future catches. Unfortunately for most fisheries, independent surveys of abundance or biomass have not been completed throughout the history of the fishery, and are often commenced after a period of rapid expansion of the fishery and consequent depletion of the population (e.g. Worthington *et al.* 2001). Surveys of abundance before, or concurrent with the development of a fishery can provide information to considerably reduce the uncertainty associated with stock assessments and the likely effects of future catches (Smith 1993, Walters 1998).

Many studies have investigated spatial and temporal variation in the density of sea urchins, but comparatively few have been designed to enable estimates of abundance or biomass over large areas. No studies that we are aware of have produced estimates of abundance or biomass of a population prior to the development of a large fishery. Indeed, independent surveys of abundance have rarely been completed at any stage of the rapid development and decline of many sea urchin fisheries around the world (Andrew *et al.* 2002). The limited development of sea urchin fisheries in NSW and eastern Victoria provides an ideal opportunity to establish accurate and precise sampling designs to provide independent estimates of abundance and biomass prior to much depletion of the population.

Here we describe the development of methods to assess the large-scale abundance and biomass of several species of sea urchin associated with commercial fisheries in NSW and eastern Victoria. In NSW, these species include the red (*Heliocidaris tuberculata*) and purple (*Centrostephanus rodgersii*) sea urchins, whilst in Victoria the species include the purple (*Centrostephanus*) and white or green sea urchin (*Heliocidaris erythrogramma*). In each case, the methods developed involve stratified, random surveys of abundance and size-structure. To enable the re-sampling of all sites involved in each of the surveys, a manual has been prepared describing their specific location and other details essential to any future survey. These manuals are available from NSW Fisheries and held at the Cronulla Fisheries Centre library.

2.2. Assessment of stocks of red sea urchins in NSW

C. Blount and D.G. Worthington

2.2.1. Introduction

During periods in the 1970s and 1980s there were attempts to develop fisheries for several species of sea urchin in NSW, but all were relatively unsuccessful. For about 15 years prior to 1998, the commercial catch of sea urchins in NSW averaged less then 2 t per year. During 1999, the commercial catch increased to 19 t, and increased again to 120 t in 2000, and 76 t in 2001. Commercial catches were dominated by the red sea urchin (*Heliocidaris tuberculata*), but also included large amounts of the purple sea urchin (*Centrostephanus rodgersii*). The two species are abundant on the mid-south coast of NSW and provided the opportunity for co-development of two fisheries for several reasons. First, the *Centrostephanus* population was apparently enormous suggesting the potential for a large fishery, but its development was limited by difficulties associated with processing roe (i.e. removal of roe, bitterness, difficulty of selling live) and a limited harvesting season. *Heliocidaris* could be harvested at times when *Centrostephanus* could not, and roe was generally of a higher quality with less variability and fewer processing problems, but clearly a more restricted and limited population. The combination of these two fisheries provided the opportunity for several efficiencies to the industry that could help facilitate their further development.

Heliocidaris is found on coastal reefs between approximately Coffs Harbour and Disaster Bay in NSW, predominantly in depths of <6-10 m. As well as forming dense aggregations in certain areas, *Heliocidaris* is also found at lower densities distributed throughout near-shore reef assemblages. Little is known about the biology and ecology of *Heliocidaris*, but Laegdsgaard and Byrne (1991) reported a protracted spawning season of nine months, between February and October. Spawning is apparently asynchronous, as some populations of red sea urchin maintain firm roe of a relatively large volume throughout the year, enabling ongoing harvesting. Laegdsgaard and Byrne (1991) also found red sea urchins to be highly fecund, producing small, planktotrophic larvae with a development time of 3 to 5 weeks.

Here we describe a survey, independent of the fishery, to provide estimates of the biomass of *Heliocidaris* within several regions on the NSW coast. The survey estimates biomass by combining information on the density and size of individuals with estimates of the area of appropriate habitat within each region. Following development of the fishery, catches were concentrated in three of the regions, and the surveys were repeated in areas open and closed to the fishery within each of these regions. This provided an estimate of the change in biomass within each region in response to known catches, and a comparison with un-fished areas.

2.2.2. Methods

2.2.2.1. Commercial catch

Commercial fishers are required to provide records of their daily catch and effort within 86 sub-zones on the NSW coast. This involves a validated total weight of sea urchins, the weight and number of individuals within each bin, and an estimate of the dive time. These records can be summed across divers and sub-zones to provide estimates of the total commercial catch and effort over larger spatial scales. Here, we summarise this catch and effort information for specific sub-zones (see Appendix 7.1) and regions of the fishery (see Figure 2.2.1).

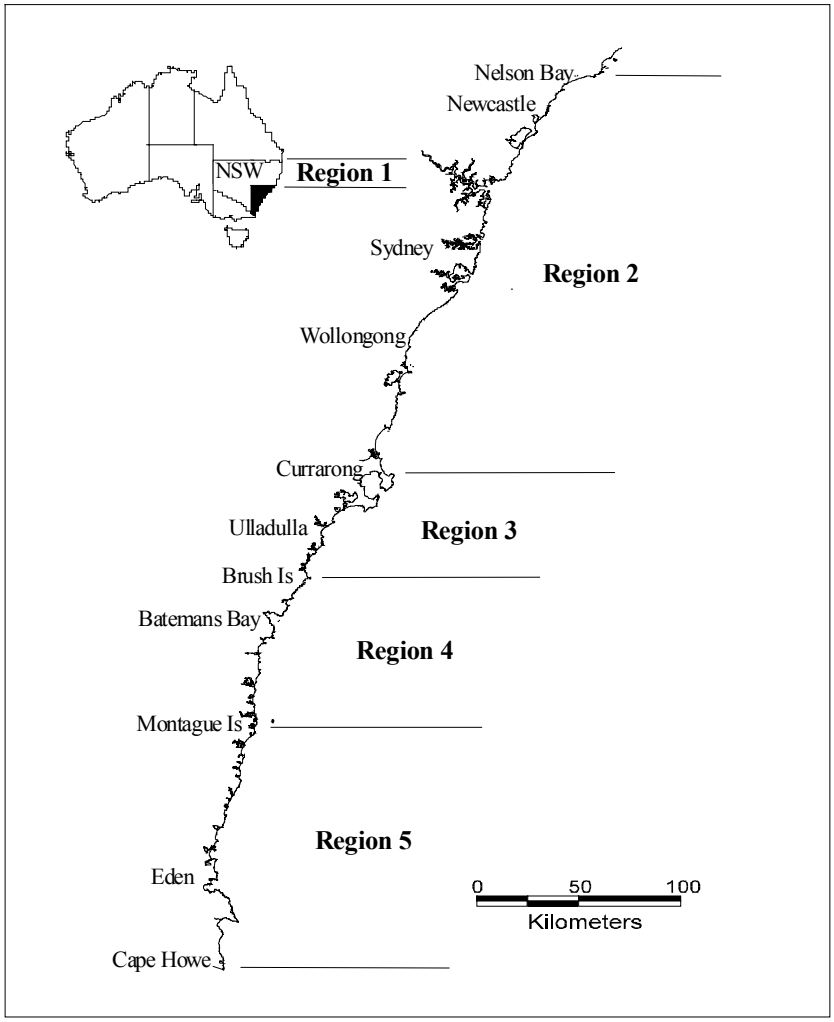

Figure 2.2.1. Map of New South Wales showing regions of the NSW Sea Urchin and Turban Shell fishery, and which were used for stock assessment of *H. tuberculata*. Major ports and break points between regions are also labeled.

2.2.2.2. Density and biomass

Surveys were designed to estimate the density of *Heliocidaris* on coastal reefs within five regions of NSW (Figure 2.2.1). The density of individuals was estimated at 10 haphazardly chosen sites within each region between Feb and Jun 2000, although the vast majority of sites were sampled between Mar and Apr 2000. Pilot studies indicated that *Heliocidaris* were abundant in all types of habitat (see Underwood *et al.* 1991) on near-shore reefs, except Barrens caused by *Centrostephanus*, where they are rare. Further, densities of *Heliocidaris* often decline with distance from the shore, and few individuals were observed beyond 50 m from the shore. As a result, at each site two tapes were placed perpendicular to the shore ~20 m apart, starting as close to shore as possible and ending 50 m from shore, or where the reef stopped or changed into Barrens. Two, 5 × 1 m transects were sampled at each of six equally spaced points spread along each tape. This involved counting the abundance of *Heliocidaris* in three size classes (i.e. small <50 mm, medium 50-100 mm, and large >100 mm). The test diameter of a sample of 30 sea urchins was also measured at each site to estimate the size-structure of individuals.

Analysis of variance was used to investigate changes in the density of the different size-classes of *Heliocidaris*. Region was considered a random factor, and Site, Tape and Transect were hierarchically nested within Region. Cochran's test was used to determine the homogeneity of variances and the data transformed where necessary. Variance components were calculated on un-transformed data and expressed as a proportion of the total variation.

An estimate of the area of appropriate habitat for *Heliocidaris* within each region was also made. This was calculated by identifying the area of reef within 50 m of shore that was not Barrens from a series of digitised aerial photographs of the NSW coast. The biomass of *Heliocidaris* within each region was then estimated by combining the area of appropriate habitat with the average density and size of sea urchins within each region. The average weight of individuals in each region was estimated from the relationship between test diameter and total weight (i.e. *Total weight (g) = 0.001647 x Test diameter (mm)* $^{2.711}$, $R^2 = 0.84$, n = 577). The standard error of the estimated biomass was calculated using,

$$SE = \sqrt{(c \times U \times V)^2 \left[\frac{\sigma_U^2}{U^2} + \frac{\sigma_V^2}{V^2} \right]}$$

where c = the area of habitat (i.e. m^2 and assumed to be known without error), U = the average density of individuals (m^{-2}), V = the average weight of individuals (g), and σ_U^2 and σ_V^2 are the corresponding variances of U and V. That is, no covariance was assumed between U and V. The area of habitat was assumed to be known without error as we had no formal quantitative estimate of error, but it is likely to be less than 10%.

2.2.2.3. Depletion caused by fishing

Sites within areas open and closed to commercial fishing were sampled both before and after the development of the fishery during 2000. This was possible within three regions (i.e. 1, 3 and 4 in Figure 2.2.1), and provided an estimate of the depletion caused by fishing. Sites were chosen to represent a range of densities of *Heliocidaris*. Some sites were the same as those used for the initial surveys of biomass, but additional sites were also sampled. The methods described above were used to estimate density at 4, 9 and 8 sites within areas open to fishing and 3, 2 and 5 sites within closed areas in Regions 1, 3 and 4, respectively. This was done in Mar-Apr 2000 and Mar-Apr 2001.

2.2.3. Results

2.2.3.1. Commercial catch

For many years prior to 1998 there was little commercial catch of *Heliocidaris* in NSW. Since 1998, the commercial catch increased rapidly and peaked in 2000, when 83 t were landed. Catches declined again in 2001 to 25 t. Most (i.e. 83-99%) of the commercial catch of *Heliocidaris* between 1998 and 2001 was taken from Regions 3 and 4 (Figure 2.2.2). Over 59 t was caught from Region 3 in 2000, and only small catches were made in all other regions (i.e. <5 t in any year). Daily catch rates ranged from 42-140 $kg.h^{-1}$ within Regions 3 and 4, with little change between 1998 and 2001. (Figure 2.2.3). During this time several sub-zones were opened to fishing (i.e. M1 and N2 in Region 3, and R1 and S1 in Region 4) and there is some evidence of serial depletion maintaining catch rates. For example, most catch from Region 3 came from subzones M2 and N3 in 1998-1999 (Figure 2.2.4). When N2 was opened early in 2000, most of the catch came from this sub-zone for the next 6 months. Since then catch in Region 3 has been more evenly spread amongst sub-zones.

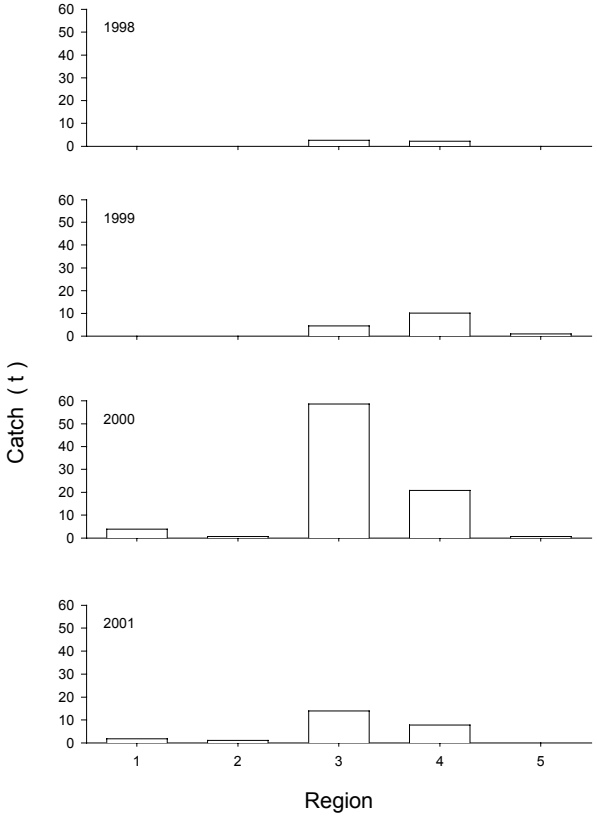

Figure 2.2.2. Commercial catch of *H. tuberculata* within five regions from 1998 to 2001.

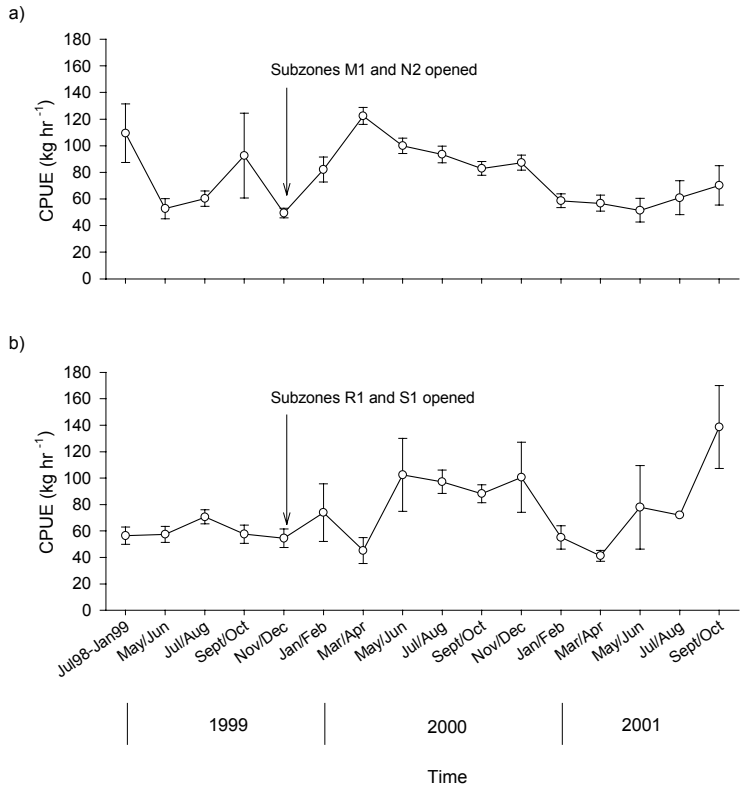

Figure 2.2.3. Commercial CPUE (± SE) of *H. tuberculata* within Region 3 from July 1998 to December 2001.

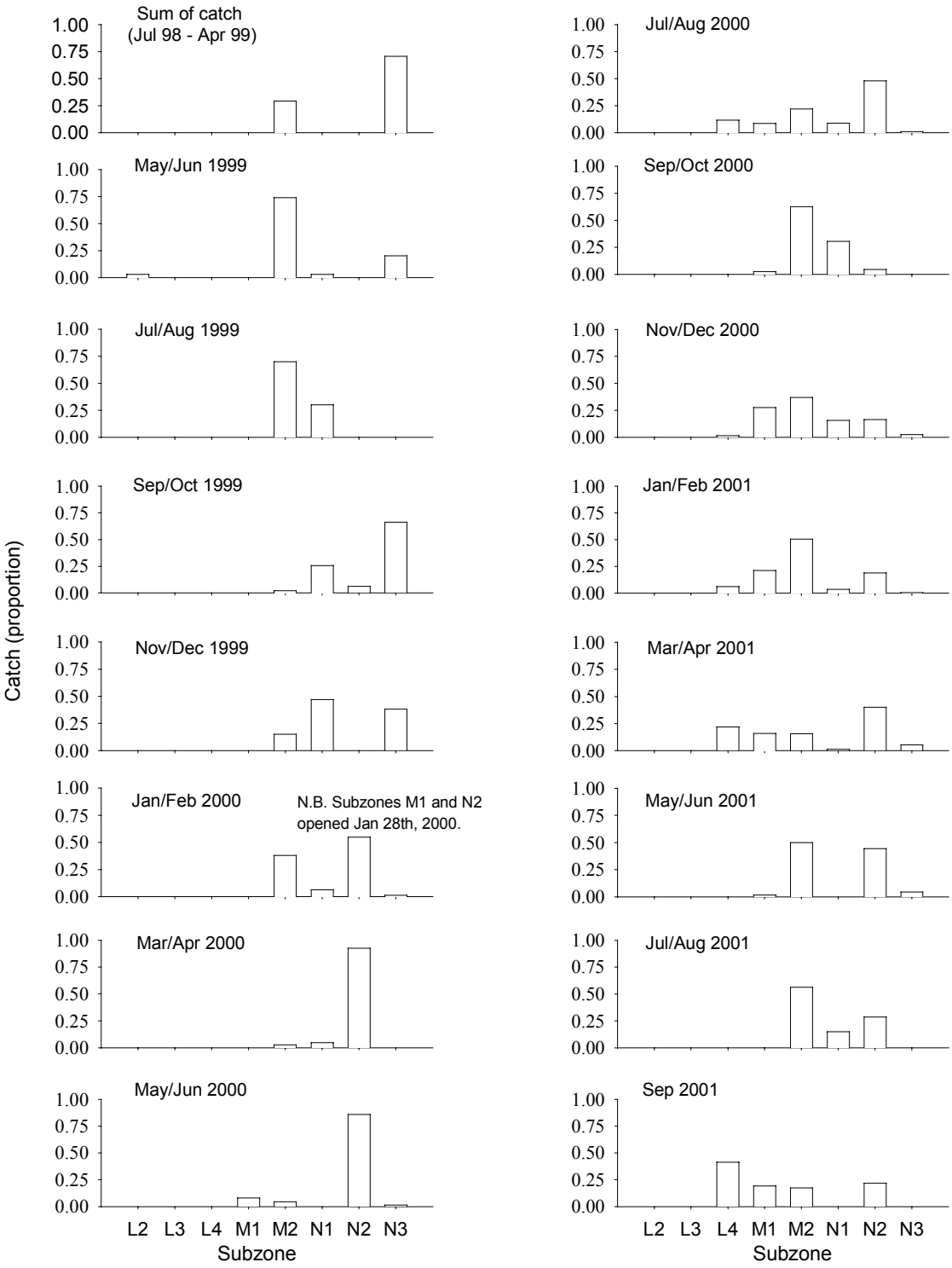

Figure 2.2.4. Proportion of bimonthly commercial catch of *H. tuberculata* in several subzones within Region 3 from July 1998 to September 2001.

2.2.3.2. Density and biomass

There was significant variation in the density of *Heliocidaris* among regions (Figure 2.2.5 and Table 2.2.1). Within regions 1-4, *Heliocidaris* was found at a range of densities (i.e. up to 8.3 per 5 m^2) at 95% of the sites sampled, but was only present within 20% of the sites sampled in Region 5, and at low densities (i.e. <0.23 per 5 m^2. Average densities were greatest within Region 3 and 4, intermediate within Region 1 and 2 and lowest in Region 5 (Figure 2.2.5, SNK test, P <0.05). Average densities were greatest within Region 3 and 4, intermediate within Region 1 and 2 and lowest in Region 5 (Figure 2.2.5, SNK test, P <0.05). There was also significant variation in density among sites within regions (Table 2.2.1), particularly in Regions 3 and 4 where some sites had very dense populations (i.e. up to 8.3 and 6.1 per 5 m^2). Most of the variation in density for small, medium, large and total individuals occurred at smaller spatial scales among transects within a tape, or among sites (Table 2.2.1).

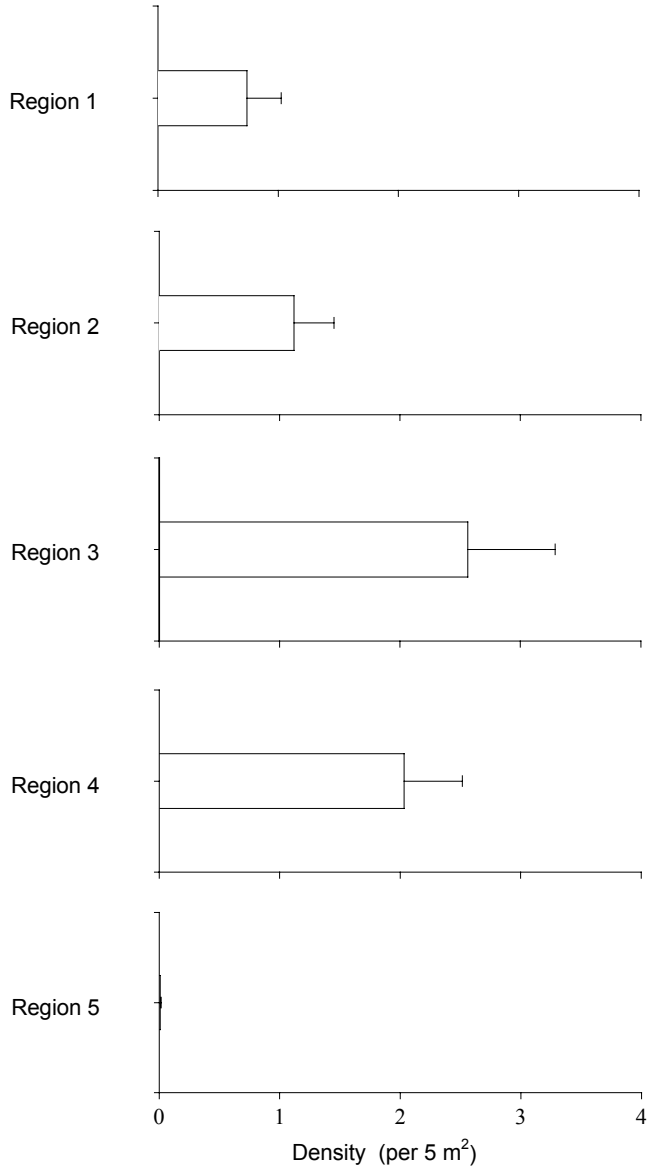

Figure 2.2.5. Density (+ SE) of *H. tuberculata* within five regions during 2000.

Table 2.2.1. Summary of analysis of variance in the density of small, medium, large and total individuals of *H. tuberculata* in New South Wales. An * shows significant effects (P <0.05).

		Small			Medium			Large			Total		
Source	**df**	**MS**	**F**	**% Var**	**MS**	**F**	**% Var**	**MS**	**F**	**% Var**	**MS**	**F**	**% Var**
Region, R	4	1.60	3.17 *	3	124.8	3.02 *	6	10.42	2.19	4	235.5	3.4 *	8
Site, S(R)	45	0.51	4.94 *	11	41.34	16.36 *	30	4.76	8.9 *	32	69.17	19.75 *	32
Tape (S)	50	0.10	0.76	0	2.53	0.75	0	0.54	1.61 *	3	3.50	0.68	0
Res	1100	0.13		86	3.39		64	0.33		60	5.17		60
Total	1199												

Medium and large individuals dominated populations at all sites. Indeed, small (<50 mm) individuals were rare, and the maximum density observed at any site was only 0.7 ± 0.2 per 5 m^2. At many sites, small individuals were not observed at all, although it is not clear to what extent this was caused by inefficient sampling. Although small individuals were rare in all regions, they were most abundant in Region 3 and 4, where the average density was 0.2 ± 0.1 and 0.1 ± 0.1 per 5 m^2 respectively.

The area of appropriate reef habitat for *Heliocidaris* ranged from as little as 163 ha in Region 3 to as much as 849 ha in Region 2 (Table 2.2.2). This is largely to do with differences between regions in the length of coastline, but also because of differences in the proportion of Barrens between regions. There were also differences in the average weight of individuals between regions (Table 2.2.2). The estimated total biomass of *Heliocidaris* within NSW was 1195 t. The largest biomass occurred in Region 2, but substantial biomass also occurred in Region 1, 3 and 4. Despite apparently appropriate habitat, the biomass in Region 5 was negligible as only very low densities were observed.

Table 2.2.2. Estimates of the density and size of individuals, and reef area with the calculated total biomass (with SE) of *H. tuberculata* within five regions in NSW.

		Region 1	**Region 2**	**Region 3**	**Region 4**	**Region 5**
Density (m^{-2})	A	0.15 (0.11)	0.22 (0.07)	0.51 (0.29)	0.41(0.19)	0.01 (0.01)
Weight (kg)	B	0.37 (0.03)	0.27 (0.04)	0.32 (0.03)	0.32 (0.02)	0.32 (0.02)
Reef area (ha)	C	283	849	163	191	303
Biomass (t)	AxBxC	154 (118)	517 (174)	265 (153)	252 (119)	7 (5)

2.2.3.3. Depletion caused by fishing

Surveys indicated significant decreases in density within areas open to commercial fishing in Regions 3 and 4, but not in Region 1 (Figure 2.2.6). In Region 3, the average density was reduced to 55% of that prior to the intense fishing. Despite that, there was substantial variation in the size of the decline amongst sites. For example, density at one site within Region 3 was depleted to 12%, whilst at another site open to fishing densities increased by 50%. A similar pattern occurred within Region 4, where density was reduced on average to 61% of that prior to the intense fishing, and their was substantial variation in the decline of density among sites. Within Region 3, reductions in density also occurred in areas closed to commercial fishing, with the average density reduced to 69%. Some of this decline is probably explained by illegal catches in the closed areas. Such declines did not occur in Region 4, where density in closed areas remained similar.

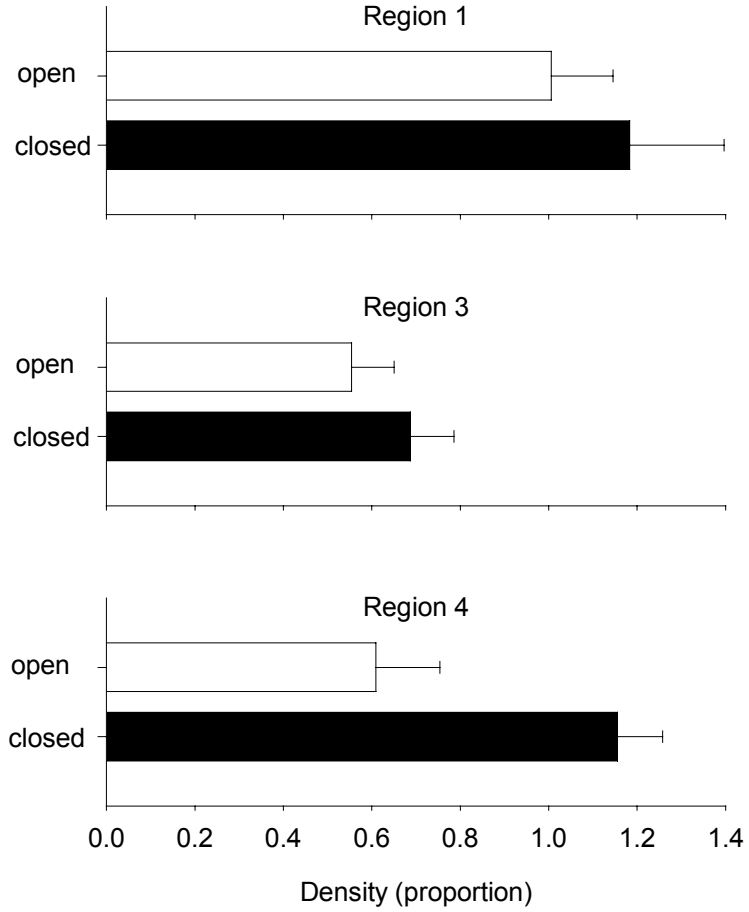

Figure 2.2.6. Depletion in density (+ SE) of *H. tuberculata* in areas open and closed to fishing within 3 regions. Depletion is expressed as relative change in density between 2000 and 2001.

When changes in density during the period of intense fishing were averaged across areas open and closed to commercial fishing, densities declined to 47% within Region 3, 82% within Region 4, and increased to 108% within Region 1. During this time, a total of 83 t of *Heliocidaris* was caught, with most coming from Region 3 and 4 (i.e. 67 t and 24 t) and little from Region 1 (i.e. 3.8 t). Assuming fishing caused all of the reduction in biomass, this provides an estimate of the total biomass of *Heliocidaris* within Region 3 and 4 prior to the intense fishing of 156 t and 126 t. These are 41% and 50% less than estimates calculated by combining average density with the area of habitat, suggesting some of the change in biomass may have been caused by factors other than the reported catches (i.e. illegal catch and recruitment).

2.2.4. Discussion

Heliocidaris was found at sites along most of the coast of NSW, from Diamond Head to south of Eden. The greatest densities occurred near the middle of this distribution from Ulladulla to Batemans Bay and Montague Island. Densities of *Heliocidaris* in areas north and south of this area tended to decrease towards the limit of the distribution. There was large variation in the density of *Heliocidaris* at a range of spatial scales. In particular, large accumulations of individuals were observed at some sites (e.g. Bawley Pt, Brush Island, Jervis Bay), whilst at other sites nearby there were few individuals. Indeed, much of the variation in density occurred at smaller spatial scales among sites or among transects.

In addition to documenting natural variation in the density of *Heliocidaris* at different spatial scales, the surveys also provide information about the effects of fishing at several spatial scales. Further, by estimating the effects of fishing, the repetition of the surveys also provided an additional method to estimate the unexploited biomass of *Heliocidaris*. The similarity of the estimates of unexploited biomass from the original surveys, and the depletion of the biomass given a known catch provide additional support for the validity of each method. This includes the aerial photographs used to estimate the area of habitat, and the representivity of the survey within the defined habitat area.

The initial concentration of the fishery in Region 3 and 4 was strongly related to the proximity of processing factories and the high density of *Heliocidaris*. In addition, at the time the fishery began to develop in 1999, many sub-zones in Region 1 and 2 were closed to commercial fishing. In the short period of intense fishing for *Heliocidaris*, concentration of effort and catch in Region 3 and 4 caused a significant depletion of the biomass (i.e. to 47% and 82%). Some sites were particularly targeted by commercial fishers, with depletions to 12-15% of the biomass within both regions. Further, the shift of effort through time suggests the serial depletion of dense aggregations of *Heliocidaris*. The rapid and large depletion of the biomass of *Heliocidaris*, combined with the low productivity of sea urchin populations, suggests the fishery should be managed with care.

Estimates of biomass and likely productivity can be used to suggest appropriate catches for fisheries (Francis 1993). During 2000 the estimates of biomass and likely productivity of the *Heliocidaris* population in NSW described above were provided to the Total Allowable Catch Setting and Review Committee. The committee concluded the large catches from Region 3 and 4 were not sustainable. They also recommended a state-wide TACC of 60 t per year for the next 5 years to be spread across regions according to the remaining biomass. The committee suggested that although this TACC was conservative, it should be enforced until more information about the productivity of the population became available. This can be investigated through ongoing monitoring of the population and experiments to investigate the response of the population to fishing.

2.3. Assessment of stocks of purple sea urchins in NSW

C. Blount and D.G. Worthington

2.3.1. Introduction

During periods in the 1970s and 1980s there were attempts to develop fisheries for several species of sea urchin in NSW, but all were relatively unsuccessful. For about 15 years prior to 1998, the commercial catch of sea urchins in NSW averaged less then 2 t per year. During 1999, the commercial catch increased to 19 t, and increased again to 120 t in 2000, and 76 t in 2001. Commercial catches were dominated by the red sea urchin (*Heliocidaris tuberculata*), but also included large amounts of the purple sea urchin (*Centrostephanus rodgersii*). The two species are abundant on the mid-south coast of NSW and provided the opportunity for co-development of two fisheries for several reasons. First, the *Centrostephanus* population was apparently enormous suggesting the potential for a large fishery, but its development was limited by difficulties associated with processing roe (i.e. removal of roe, bitterness, difficulty of selling live) and a limited harvesting season. *Heliocidaris* could be harvested at times when *Centrostephanus* could not, and roe was generally of a higher quality with less variability and fewer processing problems, but clearly a more restricted and limited population. The combination of these two fisheries provided the opportunity for several efficiencies to the industry that could help facilitate their further development.

Centrostephanus is found on shallow, rocky reefs from northern NSW to Victoria and Tasmania. Anecdotal evidence suggests *Centrostephanus* is rare west of Sandpatch Point in eastern Victoria, but is spreading down the east coast of Tasmania. As well as being found through many habitats dominated by macro-algae in shallow water, termed Fringe, *Centrostephanus* is also found in dense aggregations that maintain the reef free of macro-algae, termed Barrens (Underwood *et al.* 1991). Much is known about several aspects of the biology and ecology of *Centrostephanus*. This includes particularly its reproductive biology, age and growth and the effects of its grazing on the ecology of sub-tidal reefs in NSW (see review by Andrew and Byrne 2000). Further, the small, planktotrophic larvae of *Centrostephanus* with a development time of 3 to 5 weeks suggests the potential for the wide dispersal of this species. Indeed, although there are three separate fisheries for *Centrostephanus* in NSW, Victoria and Tasmania, there may only be a single stock.

Here we describe a survey, designed to be independent of the fishery, to provide estimates of the biomass of *Centrostephanus* in two habitats within five regions on the NSW coast. The survey estimates biomass by combining information on the density and size of individuals with estimates of the area of appropriate habitat within each region.

2.3.2. Methods

Surveys were designed to estimate the density of *Centrostephanus* on coastal reefs within five regions of NSW (Figure 2.3.1). The density of individuals was estimated at 74 haphazardly chosen sites between March and April 2000. Pilot studies indicated that the density of *Centrostephanus* varied with depth and distance from the shore. Shallow, sub-tidal rocky reefs in NSW generally consist of a range of depth-related habitats. Fringe often dominates in shallow water while slightly deeper there can be high densities of *Centrostephanus* that maintain Barrens. The interface between Fringe and Barrens often occurs at depths of about 5-10 m. As a result, at each site two tapes were placed perpendicular to the shore ~20 m apart, starting as close to shore as possible and ending 65 m from shore, or where the reef stopped. Two, 5 × 1 m transects were sampled at each of six equally spaced points spread along a tape within Fringe. When Barrens were present, two transects were sampled at three points on the tape spread at 5 m intervals from the interface

between Fringe and Barrens. This involved counting the abundance of *Centrostephanus* in three size classes (i.e. small <50 mm, medium 50-100 mm, and large >100 mm).

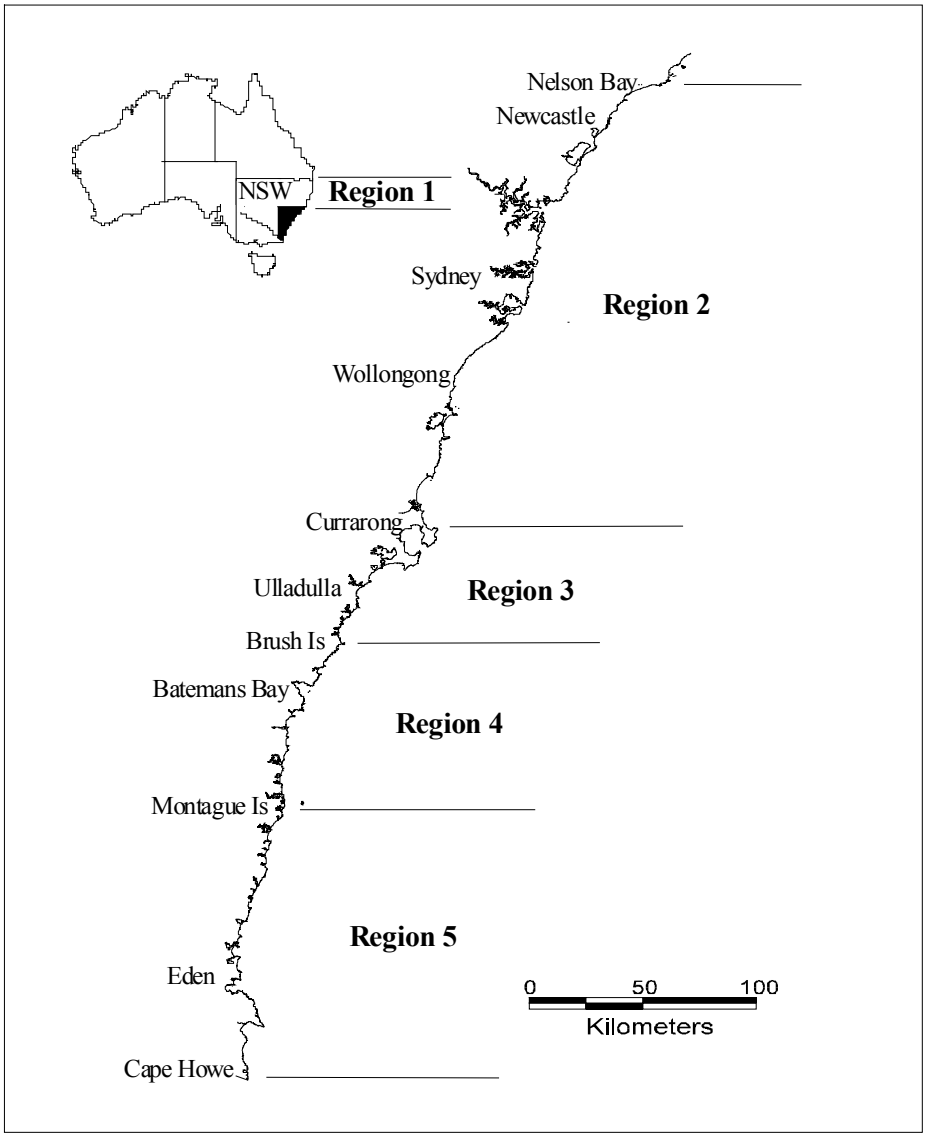

Figure 2.3.1. Map of New South Wales showing regions of the NSW Sea Urchin and Turban Shell fishery, and which were used for stock assessment of *Centrostephanus*. Major ports and break points between regions are also labeled.

At sites where *Centrostephanus* was present, divers collected six samples of seven individuals in Fringe, and three samples of seven individuals in Barrens. These samples were brought to the boat where the test diameter of all individuals was measured, and the colour of the roe graded. Roe was graded into three categories of colour (i.e. Good, Medium and Poor) related to market preference. This was done by comparing the colour of roe from each sea urchin with a standard colour chart. Sea urchins from each sample were pooled to estimate the proportion of individuals with roe of each category.

Analysis of variance was used to investigate changes in the density of the different size-classes of *Centrostephanus*. Region was considered a random factor, and Site and Tape were hierarchically nested within Region. Analysis of variance also was used to investigate changes in the proportion of individuals with roe of a Good colour. The number of sites per region was balanced for analysis

of variance by the removal of randomly selected sites where necessary. As a result, 10 sites per region for Fringe and 6 sites per region for Barrens were available for analysis. Cochran's test was used to determine the homogeneity of variances and the data transformed where necessary. Variance components were calculated on un-transformed data and expressed as a proportion of the total variation.

Estimates of the area of Fringe and Barrens within each region were made by identifying these habitats in a series of digitised aerial photographs of the coast of NSW. Estimates were restricted to 100 m from the shore because of difficulties interpreting aerial photographs in deeper water. The biomass of *Centrostephanus* within Fringe and Barrens could then be calculated by combining estimates of average density and weight of individuals, and the area of habitat within each region. The standard error of the estimated biomass was calculated using a formula for products of variances (see Section 2.2.2.2). The average weight of individuals in each region was estimated using the relationship between test diameter and total weight of individuals for each species (i.e. $Total\ weight\ (g) = 0.002382 \times Test\ diameter\ (mm)^{2.651}$, $R^2 = 0.86$, $n = 445$).

2.3.3. Results

There was no significant difference in the density of *Centrostephanus* among regions (Figure 2.3.2 and Table 2.3.1). *Centrostephanus* was found at a range of densities (i.e. up to 34.6 per 5 m^2 in Fringe and 56.0 per 5 m^2 in Barrens) at 96% of the sites sampled. Average densities were higher in Barrens than Fringe (Figure 2.3.2, SNK test, $P < 0.05$). There was also significant variation in density among sites within regions. Most of the variation in density for small, medium, large and total individuals occurred at smaller spatial scales among transects within a tape, or among sites (Table 2.3.1). Individuals in the medium size class dominated the population at all sites (Figure 2.3.2). Indeed, small (<50 mm) individuals were not abundant, with about 1.0 per 5 m^2. It is not clear to what extent this was caused by inefficient sampling of this size class. The size structure of *Centrostephanus* in Fringe was often different from Barrens (Figure 2.3.4). More large individuals were found in Fringe than Barrens at most sites.

There was no significant variation among regions in the proportion of individuals with roe of a Good colour in either Fringe or Barrens. There was significant variation in the proportion of individuals with roe of a Good colour among sites (Figure 2.3.3, Table 2.3.2). Most of the variation in the proportion of individuals with roe of a Good colour occurred at smaller spatial scales among samples within sites and among sites.

The area of reef with Fringe and Barrens ranged from 259 ha and 248 ha in Region 3 to 1505 ha and 632 ha in Region 2 (Table 2.3.3). As the average density and weight of individuals were similar among regions, differences in the area of reef dominated the differences in biomass among regions. The estimated total biomass of *Centrostephanus* within NSW was about 52 000 t, with about 22 000 t in Fringe and about 30 000 t in Barrens. In both of these habitats, most of the biomass occurred within Region 2.

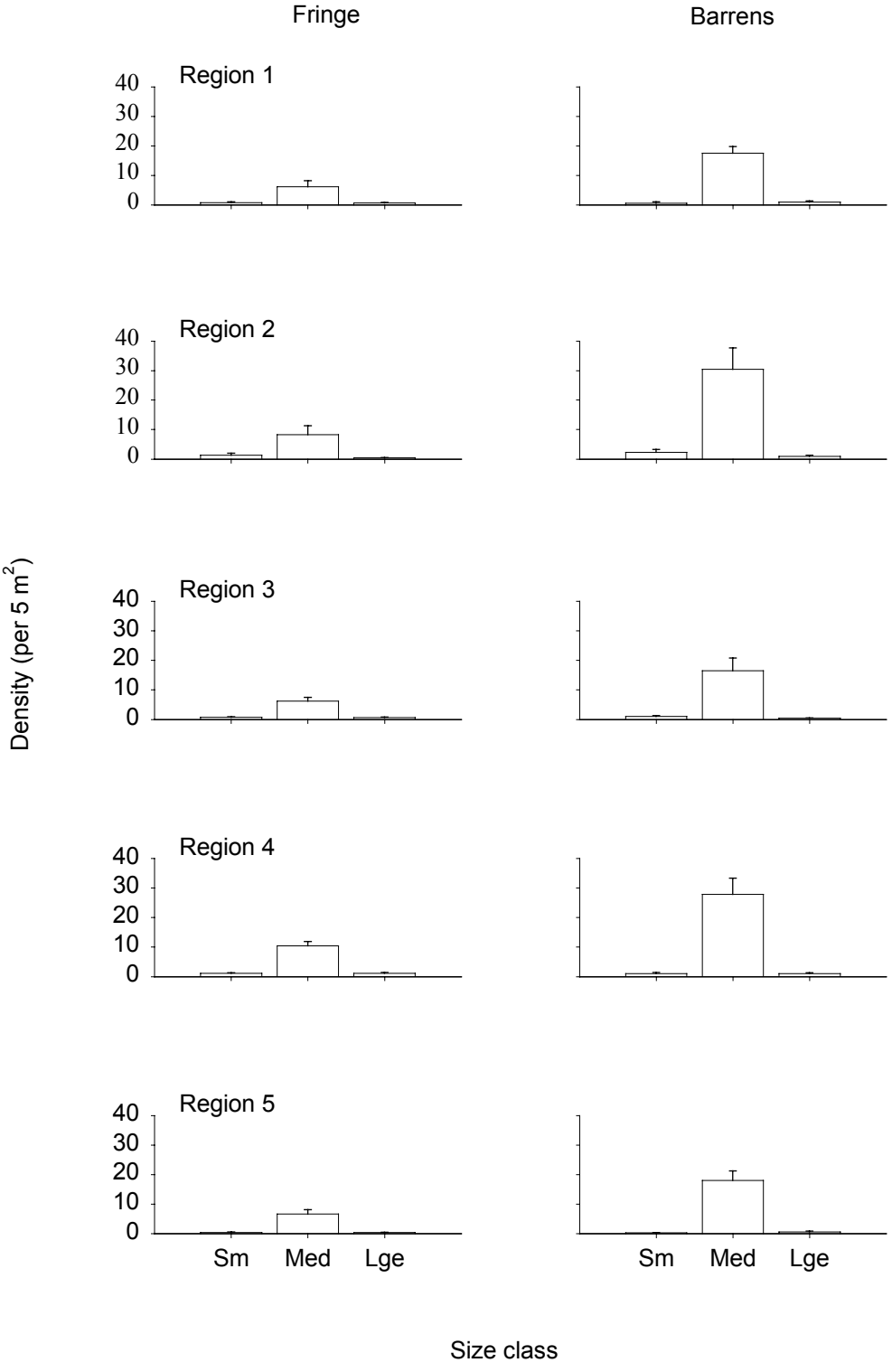

Figure 2.3.2. Density (+ SE) of small, medium and large *Centrostephanus* in Fringe and Barrens within five regions during 2000.

Table 2.3.1. Summary of analysis of variance in the density of small, medium, large and total individuals of *Centrostephanus* in a) Fringe and b) Barrens. An * shows significant effects (P <0.05).

a) Fringe		Small			Medium			Large			Total		
Source	**df**	**MS**	**F**	**% Var**	**MS**	**F**	**% Var**	**MS**	**F**	**% Var**	**MS**	**F**	**% Var**
Region, R	4	29.95	1.06	0	709.81	0.75	0	18.56	2.356	0	1125.94	0.95	0
Site, S(R)	45	28.22	4.68 *	22	940.58	9.51 *	38	7.88	3.131 *	11	1190.46	9.78 *	37
Tape, T(S)	50	6.04	2.05 *	6	98.86	1.81 *	4	2.52	1.403 *	3	121.72	1.70 *	3
Res	1100	2.94		71	54.66		59	1.79		86	71.76		60
Total	1199												

b) Barrens		Small			Medium			Large			Total		
Source	**df**	**MS**	**F**	**% Var**	**MS**	**F**	**% Var**	**MS**	**F**	**% Var**	**MS**	**F**	**% Var**
Region, R	4	38.47	1.54	3	3008.30	1.81	8	4.53	0.65	0	3691.67	2.14	10
Site, S(R)	25	25.04	2.97 *	20	1663.81	14.40	53	6.95	3.81 *	22	1727.55	11.74	50
Tape, T(S)	30	8.44	1.76 *	9	115.54	1.23	1	1.83	1.22	3	147.20	1.50	3
Res	300	4.79		69	93.60		38	1.50		76	98.39		37
Total	359												

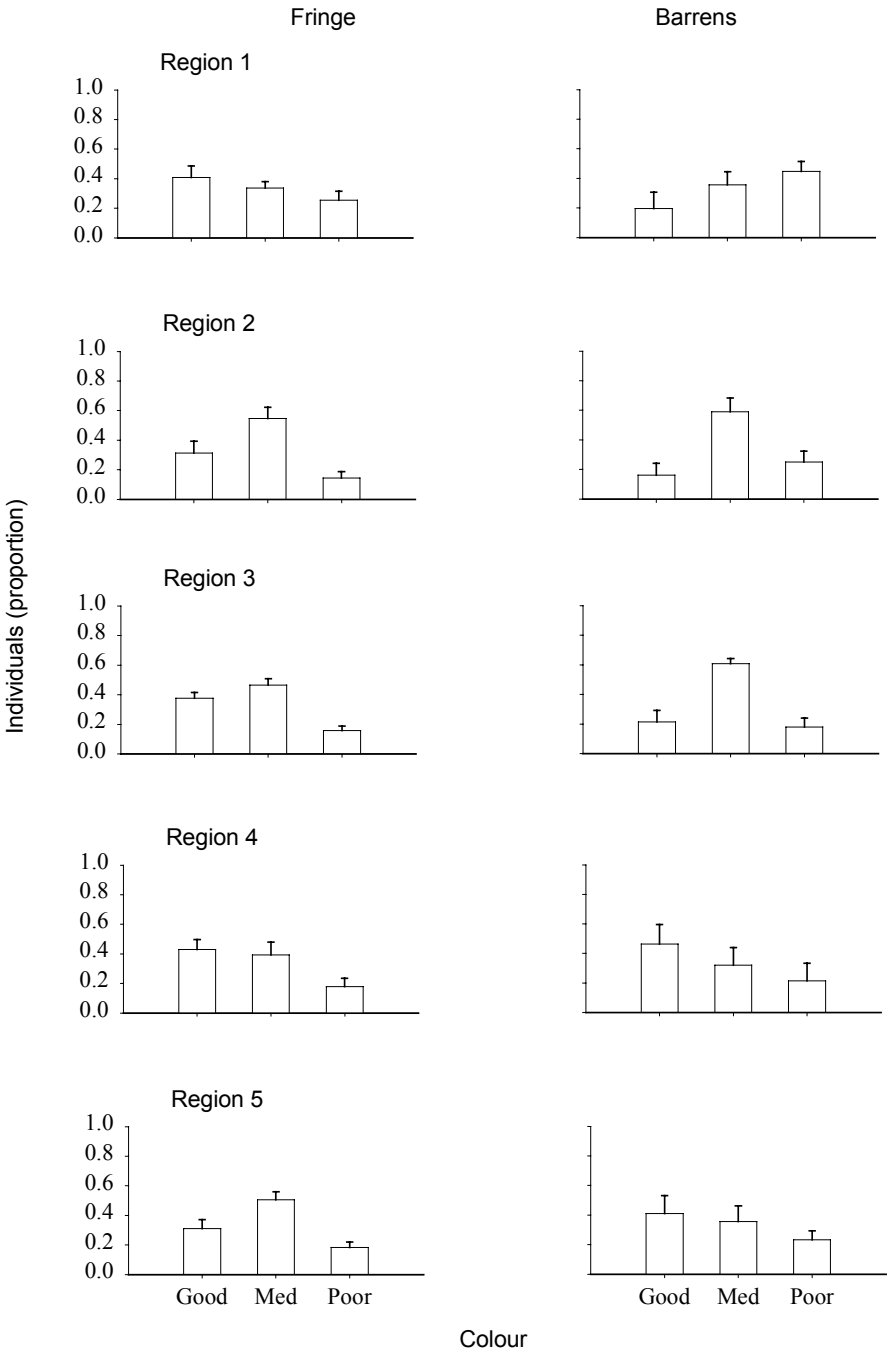

Figure 2.3.3. Proportion (+ SE) of individuals with roe of Good, Medium and Poor colour in Fringe and Barrens with five regions during 2000.

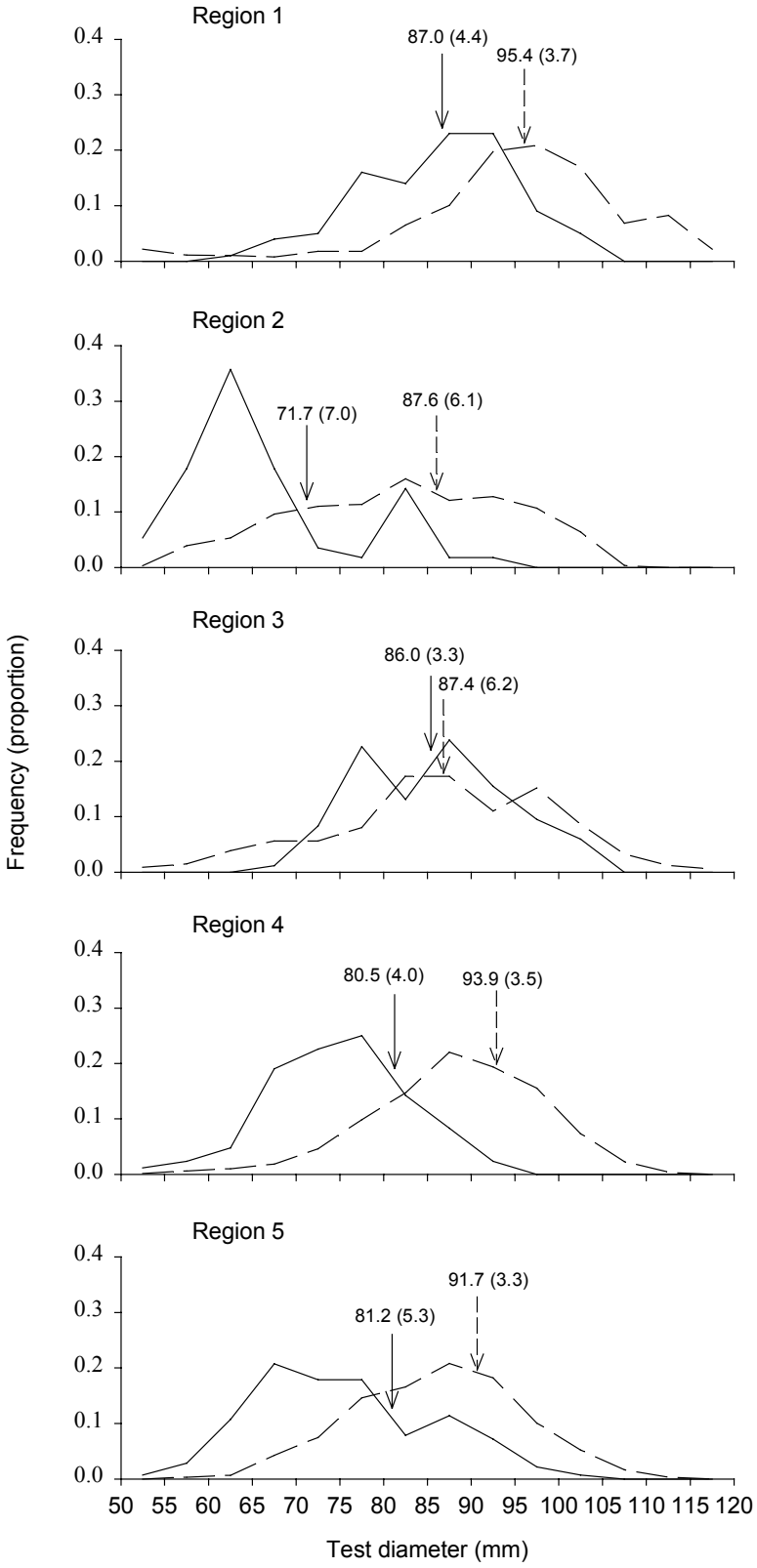

Figure 2.3.4. Size structure of *Centrostephanus* in Fringe (broken line) and Barrens (solid line) within five regions during 2000. The average test diameter (with SE) for Fringe and Barrens is also shown for each region.

Table 2.3.2. Summary of analysis of variance in the proportion of individuals with roe of good colour. An * shows significant effects (P <0.05), and [a] = arcsine transformation.

		Fringe [a]				Barrens [a]		
Source	**df**	**MS**	**F**	**% Var**	**df**	**MS**	**F**	**% Var**
Region, R	4	0.16	0.78	3	4	0.30	1.76	3
Site, S(R)	30	0.20	2.85 *	34	15	0.17	2.91 *	33
Res	105	0.07		63	20	0.06		63
Total	139				39			

Table 2.3.3. Estimates of the density and size of individuals, and reef area with the calculated total biomass (with SE) of *Centrostephanus* within Fringe and Barrens in five regions in NSW. Note, estimates are restricted to areas within 65 m from shore.

Fringe		**Region 1**	**Region 2**	**Region 3**	**Region 4**	**Region 5**
Density (m^{-2})	A	1.46 (0.63)	2.02 (0.96)	2.16 (0.80)	2.76 (0.59)	1.47 (0.51)
Weight (kg)	B	0.44 (0.02)	0.35 (0.03)	0.35 (0.02)	0.41 (0.01)	0.39 (0.01)
Reef area (ha)	C	428	1505	259	353	484
Biomass (t)	AxBxC	2735 (1192)	10667 (5117)	1943 (723)	4017 (873)	2756 (973)
Barrens						
Density (m^{-2})	A	3.82 (1.08)	6.73 (1.87)	3.57 (1.09)	5.96 (1.40)	3.80 (0.94)
Weight (kg)	B	0.33 (0.02)	0.21 (0.04)	0.33 (0.02)	0.27 (0.01)	0.28 (0.03)
Reef area (ha)	C	183	632	248	442	442
Biomass (t)	AxBxC	2303 (674)	8774 (2945)	2899 (900)	7217 (1744)	4781 (1271)

2.3.4. Discussion

Centrostephanus was found at sites along most of the coast of NSW, from Diamond Head to south of Eden. Although there was a trend of lower densities towards the edges of this distribution, there was no significant variation in density among the regions. Andrew and Underwood (1989) also found little variation in density of *Centrostephanus* between sites, although their study was restricted to Barrens at a small number of sites. As was the case for *Heliocidaris* (see Chapter 2.2) most of the variation in density occurred at smaller spatial scales among sites or among transects. The density of *Centrostephanus* in Fringe was about half that in Barrens. Further, the average size of individuals in Barrens was generally much less than in Fringe. Fringe is generally shallower and more exposed to wave action than Barrens (Underwood *et al.* 1991, Andrew and O'Neill 2000). Further, the average size of individuals in Barrens was generally much less than in Fringe.

There was no significant variation among regions in the proportion of individuals with roe of a good colour. Most variation occurred at smaller scales among samples within sites and among sites. In Fringe, most individuals had roe of a good or medium colour, while in the Barrens most individuals had roe of a medium or poor colour. Variation in the quality of roe among individuals is common in sea urchins (Andrew *et al.* 2002), and is probably related to their supply of food. The

ability to identify individuals with high quality roe could prevent a great deal of waste and increase efficiency, and can be done using a variety of indicators (Blount and Worthington, 2002). At least initially, only the sea urchins in Fringe are likely to be caught by the fishery, as yield and colour of roe from individuals in Barrens are poor (see Blount and Worthington 2002). This may change as the fishery develops and increases its use of the population in Barrens.

In addition to documenting natural variation in the density of *Centrostephanus* at different spatial scales, the surveys also provided a method for estimating the unexploited biomass of *Centrostephanus*. In Chapter 2.2, for *Heliocidaris*, this method provided similar estimates of unexploited biomass to another method involving the depletion of the biomass of *Heliocidaris*. This supports the validity of this method, including the representivity of the survey within the defined habitat area and the use aerial photographs to estimate the area of habitat.

Estimates of the biomass of *Centrostephanus* can be combined with estimates of the likely productivity of the population to calculate potentially sustainable catches. Likely productivity can be estimated from rates of growth and mortality together with an assumed relationship between stock and recruitment (Francis 1993). Deterministic estimates of Maximum Sustainable Yield (MSY) for *Centrostephanus* suggest annual catches of 1-5% of the unexploited biomass (Blount and Worthington, unpublished data). The estimated biomass in Fringe of about 20 000 t is close to the unexploited level because of limited fishing, and suggests catches of 200-1000 could be sustainable independent of the population in Barrens. However, a significant proportion (i.e. ~20%) of this biomass will contain individuals with roe of a quality that is not marketable. Similar to the fishery for red sea urchins, such catches would need to be spread along the coast in proportion to the available biomass. As recommended by the TAC Setting and Review Committee for red sea urchins, conservative catches should be taken until more is known about how the stock responds to depletion. For *Centrostephanus*, some information is already available as to how the stock may respond to reductions in density resulting from fishing. For example, Blount *et al.* (Chapter 3.2) have shown that recruitment of *Centrostephanus* may increase when densities are reduced, in contrast to what is known for some other species of sea urchin where populations show depensatory responses to harvesting (Tegner and Dayton 1977). In addition, Blount *et al* (Chapters 3.3, 3.4) have shown that reductions in density of *Centrostephanus* in Barrens, or transplanting individuals from Barrens to Fringe, can be a cost-effective method of utilising the population in Barrens. Further, the biomass of *Centrostephanus* in Barrens is likely to be much greater than the 30 000 t estimated, because of the underestimation of the area of Barrens associated with only calculating areas within 100 m from shore. Regardless, if the fishery for *Centrostephanus* were to expand, regular monitoring of the population should continue.

2.4. Assessment of stocks of sea urchins in Port Phillip Bay and eastern Victoria

C. Blount, D.G. Worthington, H. Gorfine, R.C. Chick, C. Dixon and B.R. Stewart

2.4.1. Introduction

There are several fisheries for sea urchins in Victoria. The purple sea urchin (*Centrostephanus rodgersii*) is caught in eastern Victoria, and the smaller green or white sea urchin (*Heliocidaris erythrogramma*) is caught in eastern Victoria and Port Phillip Bay. The commercial catch of *Heliocidaris* and *Centrostephanus* in these fisheries has been small (i.e. <20 t yr^{-1}) over the last five years. Anecdotal evidence suggests the population of *Centrostephanus* is much larger than that of *Heliocidaris* in eastern Victoria, and the complimentary spawning seasons could help facilitate the development of both fisheries. Anecdotal information also suggested there was a large population of *Heliocidaris* within Port Phillip Bay, but much of this population may have roe of variable quality.

Much is known about several aspects of the biology and ecology of *Heliocidaris* and *Centrostephanus*. Both *Heliocidaris* and *Centrostephanus* are found on coastal reefs in eastern Victoria, predominantly in depths of <6-10 m in habitat dominated by macro-algae. Both species are also abundant in NSW. As well as being found through many habitats dominated by macro-algae in shallow water, both species are able, at high densities, to form areas devoid of macro-algae termed Barrens (Fletcher 1987, Sanderson *et al.* 1996). The reproductive biology of *Heliocidaris* and *Centrostephanus* is quite different. *Centrostephanus* spawns in winter and *Heliocidaris* in summer. Following spawning roe recovers with firm, nutritive material being deposited prior to gametogenesis (Laegdsgaard and Byrne 1991, Byrne *et al.* 1998). As a result, *Heliocidaris* is caught as the roe is recovering mostly between May and December, while *Centrostephanus* is mostly caught between November and May. There are also differences in the dispersive potential of larvae. The small, planktotrophic larvae of *Centrostephanus* with a development time of 3 to 5 weeks suggests the potential for the wide dispersal of this species (King *et al.* 1994), while the short larval life of *Heliocidaris* suggests populations may be much more localised.

Here we describe a survey, independent of the fisheries, to provide estimates of the biomass of *Centrostephanus* and *Heliocidaris* in eastern Victoria, and *Heliocidaris* within Port Phillip Bay. The survey estimates biomass by combining information on the density and size of individuals with estimates of the area of appropriate habitats.

2.4.2. Methods

2.4.2.1. Port Phillip Bay

Surveys were designed to estimate the density of *Heliocidaris* on coastal reefs in six locations within Port Phillip Bay. Variation in the density of individuals was estimated at a range of hierarchically nested spatial scales within each location during May 2002. Each location contained large areas of solid and broken reef interspersed with areas of seagrass and sand (Figure 2.4.1a). Within each location, two reefs, consisting of continuous or broken areas of rocky substrate interspersed with areas of sand, were sampled. Within each reef, two sites separated by ~500 m were sampled. Within each site, two sub-sites separated by ~100 m were sampled. Within each sub-site, two drops separated by ~10 m were sampled. Within each drop five, 5 × 1 m transects were placed haphazardly on the reef and sea urchins within two size classes (Small <35 mm and Large >35 mm) were counted.

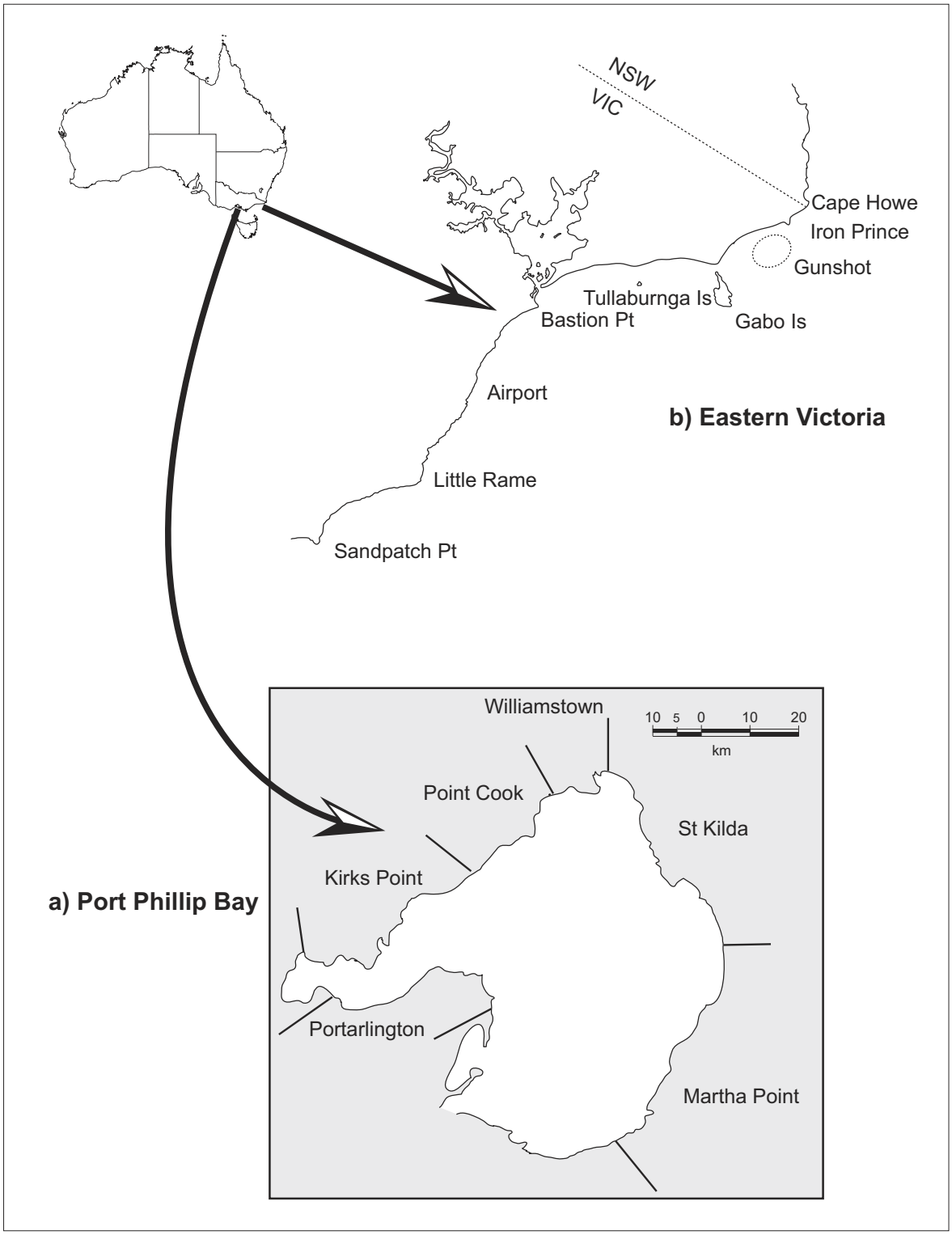

Figure 2.4.1. Map showing locations used for stock assessment in a) Port Phillip Bay and b) eastern Victoria.

At each drop, divers also collected 10 sea urchins, which were brought to the boat where their test diameter was measured. Roe from each sea urchin was assessed visually, and graded into three categories of yield and colour using a standard colour chart. For yield of the roe, the categories were Good (gonad weight as a proportion of total weight from 15-25%), Medium (5-15%) and Poor (0-5%). For colour of the roe, the categories were Good (bright yellow/orange), Medium (pale yellow/orange) and Poor (light/dark brown).

Analysis of variance was used to investigate changes in the density of the different size-classes of *Heliocidaris*. Location was considered a random factor, and Reef, Site, Sub-site and Drop were hierarchically nested within Location. Analysis of variance also was used to investigate changes in the proportion of individuals with roe of a Good colour. Cochran's test was used to determine the homogeneity of variances and the data transformed where necessary. Variance components were calculated on un-transformed data and expressed as a proportion of the total variation. Standard cost-benefit procedures were used to calculate the relative amount of replication appropriate at each spatial scale, and involved combining estimates of variation in density at different spatial scales with the marginal cost of sampling (Underwood 1981).

2.4.2.2. Eastern Victoria

Surveys were designed to estimate the density of *Heliocidaris* and *Centrostephanus* on coastal reefs within nine sites from Cape Howe to Sandpatch Point (Figure 2.4.1b). These sites are within nine of the ten large, isolated areas of reef habitat along this section of coastline. For comparison, surveys followed the same general methods as those used in NSW (see Section 2.2.2.2), and were completed in August 2000. That is, at each site two tapes were placed perpendicular to the shore and ~20 m apart, starting as close to shore as possible and ending 65 m from shore, or where the reef stopped or changed into Barrens. Two, 5 × 1 m transects were sampled at each of six equally spaced points spread along each tape. This involved counting the abundance of *Heliocidaris* and *Centrostephanus* in three size classes (i.e. small <50 mm, medium 50-100 mm, and large >100 mm). The test-diameter of a sample of 30 sea urchins was also made at each site to estimate the size-structure of individuals.

At sites where each species was present, divers also collected four samples of seven individuals at each site. These samples were brought to the boat where their test diameter of all individuals was measured, and the colour of their roe graded into three categories of colour (Good, Medium and Poor). This was done by comparing the colour of roe from each sea urchin with a standard colour chart. Sea urchins from each sample were pooled to estimate the proportion of individuals with roe of each category.

Analysis of variance was used to investigate changes in the density of the different size-classes of *Centrostephanus*. Location was considered a random factor, and Tape was nested within Location. Analysis of variance also was used to investigate changes in the proportion of individuals with roe of a Good colour. Cochran's test was used to determine the homogeneity of variances and the data transformed where necessary. Variance components were calculated on un-transformed data and expressed as a proportion of the total variation.

2.4.2.3. Habitat area and biomass

Estimates of the area of habitat available to sea urchins in Port Phillip Bay and eastern Victoria were obtained from independent sources. In Port Phillip Bay *Heliocidaris* were observed to inhabit two different types of habitat, bare reef (barrens) and reef with macro-algae. Calculations of the area of each of these habitats within Port Phillip Bay were made using data from Blake and Ball (2001). In eastern Victoria bare reef, or barrens, was rare and calculations of the area of habitat available to sea urchins were taken from estimates of reef area made by McShane *et al.* (1986). No

estimates of error were provided with these estimates, so we calculated estimates of biomass assuming they were measured without error.

For each Location in Port Phillip Bay and eastern Victoria the biomass of *Heliocidaris* and *Centrostephanus* was calculated by combining estimates of density, the weight of individuals and the area of habitat available within each location. In Port Phillip Bay, where calculations were made for two habitats, post-stratification was used to estimate the density and weight of individuals within each habitat (i.e. sites were haphazardly selected, then classified to one of the habitat types). The average weight of individuals in each location in Port Phillip Bay and eastern Victoria was estimated using the relationship between test diameter and total weight of individuals for each species. For *Heliocidaris* this was *Total weight (g) = 0.0002601 x Test diameter (mm)* $^{3.088}$ (R^2 = 0.96, n = 81), and for *Centrostephanus* this was *Total weight (g) = 0.002382 x Test diameter (mm)* $^{2.651}$ (R^2 = 0.86, n = 445). The standard error of the estimated biomass for *Centrostephanus* and *Heliocidaris* in eastern Victoria was calculated using the formula for products of variances described in Chapter 2.2, but for *Heliocidaris* in Port Phillip Bay this was modified to include covariation between density and weight of individuals,

$$SE = \sqrt{(c \times U \times V)^2 \left[\frac{\sigma_U^2}{U^2} + \frac{\sigma_V^2}{V^2} + \frac{2\,\mathrm{cov}_{UV}}{UV} \right]}$$

where c = the area of habitat (i.e. m^2 and assumed to be known without error), U = the average density of individuals (m^{-2}), V = the average weight of individuals (g), σ_U^2 and σ_V^2 are the corresponding variances of U and V, and cov_{UV} is the covariance.

2.4.3. Results

2.4.3.1. Port Phillip Bay

There was significant variation in the density of *Heliocidaris* at several spatial scales (Table 2.4.1). Variation in the density of small individuals was dominated by differences among Drops and Reefs, while variation in the density of large individuals was dominated by differences among Reefs and Locations. For example, the density of large sea urchins at Kirks Point was significantly higher than all other locations (Figure 2.4.2, SNK test, $P < 0.05$). Individuals in the large size class dominated the population at all sites (Figure 2.4.2). Indeed, small individuals were not abundant at any site, and this was not likely to be caused by inefficient sampling. Observations on the size structure of *Heliocidaris* indicated there were differences among habitats and this was also related to density. When habitats were pooled there was a significant relationship between the density and size of individuals, with smaller average sizes in sites where there were higher densities (Figure 2.4.3; linear regression, $P < 0.05$).

Differences in the yield and colour of roe among locations were related (Figure 2.4.2). That is, in locations where yield was Good, the colour of the roe was also Good. For example, at Williamstown, the proportion of individuals where the yield of roe was Good, Medium and Poor was 0.08, 0.41 and 0.51, while for colour these values were 0.17, 0.31 and 0.53. In contrast, individuals with roe of a Good yield and colour dominated populations within Point Cook and Martha Point. Variation in the proportion of individuals with roe of Good yield and colour was dominated by differences at larger spatial scales, particularly among locations for yield (Table 2.4.1).

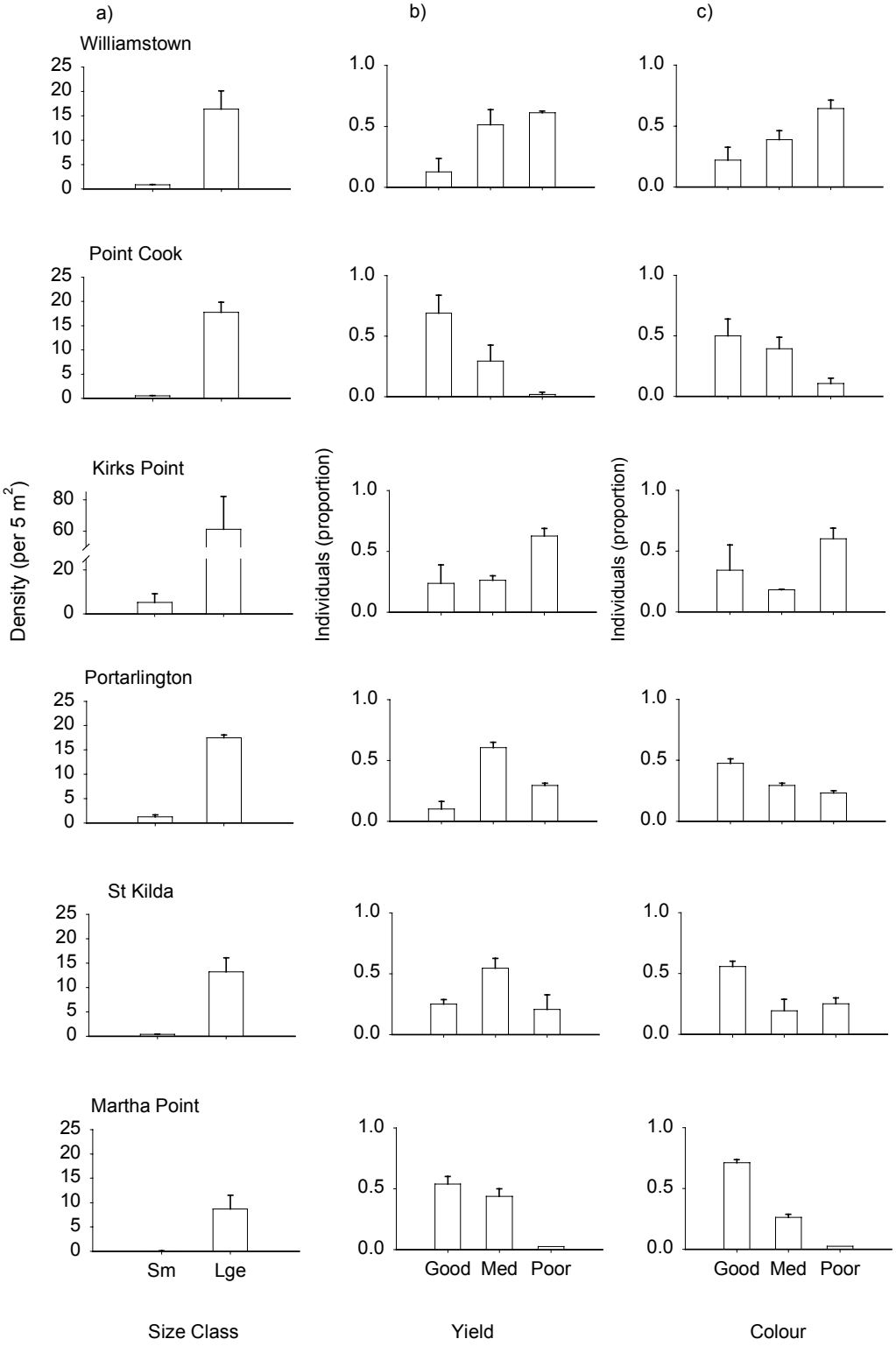

Figure 2.4.2. a) Density (+ SE) of small and large individuals, b) proportion (+ SE) of individuals with roe of Good, Medium and Poor yield, and c) proportion (+ SE) of individuals with roe of Good, Medium and Poor colour for *H. erythrogramma* within six locations during 2002.

Table 2.4.1. Summary of analysis of variance in the density of small and large individuals, yield and colour of roe of *H. erythrogramma* in Port Phillip Bay. An * shows significant effects (P <0.05), and [a] = arcsine transformation.

Density	**Small**				**Large**			
Source	**df**	**MS**	**F**	**% Var**	**df**	**MS**	**F**	**% Var**
Location, L	5	409.49	2.43	7	5	32299.80	6.03 *	46
Reef, R(L)	6	295.93	5.92 *	29	6	7589.23	1.86	22
Site, S(R)	12	62.94	1.30	0	12	1745.19	4.55 *	9
Subsite, SS(S)	24	63.76	1.32	10	24	496.21	1.77	2
Drop, D(SS)	48	44.56	4.41 *	42	48	381.46	3.15 *	9
Res	384	2.48		12	384	75.52		11
Total	479				479			
Roe	**Yield [a]**				**Colour [a]**			
Source	**df**	**MS**	**F**	**% Var**	**df**	**MS**	**F**	**% Var**
Location, L	5	1.70	5.27 *	58	5	0.95	3.59	32
Reef, R(L)	6	0.32	2.55	11	6	0.26	1.30	37
Site, S(R)	12	0.12	1.73	19	12	0.20	1.75	4
Subsite, SS(S)	24	0.07	1.33	9	24	0.11	3.73 *	19
Res	48	0.05		4	48	0.03		8
Total	95				95			

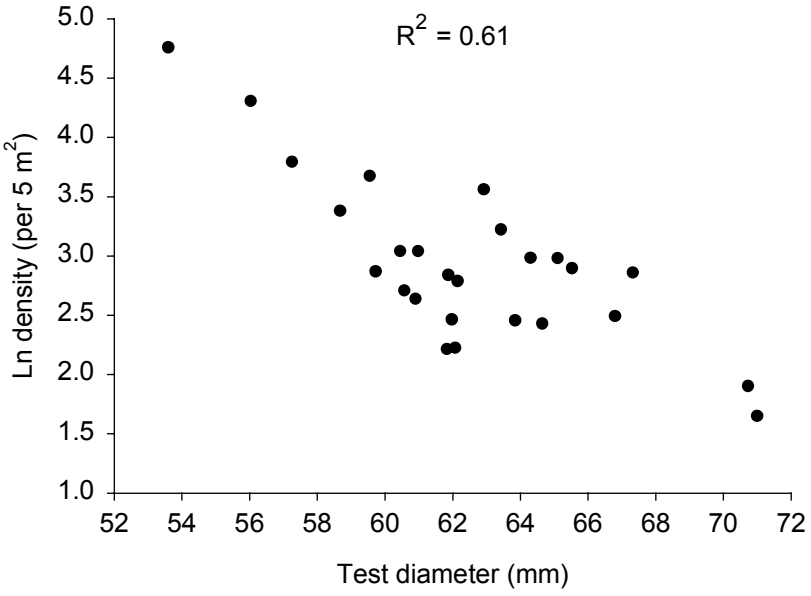

Figure 2.4.3. Relationship between mean test diameter and density of *H. erythrogramma* at sites in Port Phillip Bay during 2002.

Table 2.4.2. Estimates of the density and size of individuals, and reef area with the calculated biomass (with SE) of *H. erythrogramma* in two habitats from six locations in Port Phillip Bay. Note, estimates of reef area for Portarlington from Blake and Ball (2001) did not appear to be available during the surveys.

		Williamstown	**Pt Cook**	**Kirks Pt**	**Portarlington**	**St Kilda**	**Martha Pt**
Reef with macroalgae							
Density (m^{-2})	A	2.57 (0.54)	3.39 (0.55)	14.83 (0.72)	3.56 (0.38)	1.27 (0.67)	1.55 (0.27)
Weight (kg)	B	0.09 (0.01)	0.09 (0.01)	0.06 (0.02)	0.11 (0.01)	0.10 (0.01)	0.13 (0.01)
Reef area (ha)	C	307	504	33	456	84	160
Biomass (t)	AxBxC	711 (152)	1501 (250)	320 (94)	1829 (242)	107 (59)	317 (63)
Bare reef							
Density (m^{-2})	A	4.62 (1.32)	5.02 (0.74)	13.40 (3.38)	3.95 (0.28)	4.18 (0.44)	2.35 (0.13)
Weight (kg)	B	0.09 (0.01)	0.10 (0.01)	0.07 (0.01)	0.10 (0.01)	0.09 (0.01)	0.12 (0.02)
Reef area (ha)	C	76	18	322	0.05	212	52
Biomass (t)	AxBxC	317 (92)	90 (13)	3035 (814)	~0	780 (104)	113 (16)

The area of different habitats varied among locations (Table 2.4.2). Some locations were dominated by reef with macro-algae (e.g. Pt Cook), whilst others were dominated by bare reef (e.g. Kirks Pt). Some differences were observed between the habitat available to *Heliocidaris* during the surveys compared to those described in Blake and Ball (2001). For example, significant sand movement within Portarlington appeared to have significantly reduced the area of reef with macro-algae. This decline also probably effected the population of *Heliocidaris*, with a major decline in biomass. Individuals within the bare reef habitat tended to occur at higher densities, but at lower average weights than in the macro-algal habitat. The estimated total biomass of *Heliocidaris* in Port Phillip Bay was about 9000 t (Table 2.4.2). Of this, about 4800 t was estimated to occur on reef with macro-algae, although the biomass for Portarlington may be greatly over-estimated (see above). About 4300 t was estimated to occur on bare reef. Further, approximately 14% of the biomass on reef with macroalgae, and 45% on bare reef, were composed of individuals with poor quality roe.

2.4.3.2. Eastern Victoria

There was significant variation in the density of *Centrostephanus* and *Heliocidaris* at several scales. Despite that, variation in density of both species was dominated by differences at a small spatial scale among transects within tapes (Table 2.4.3). Densities of *Centrostephanus* were greater than *Heliocidaris* in all locations, apart from Airport and Iron Prince (Figure 2.4.4). For example, *Centrostephanus* was six times as dense as *Heliocidaris* at Sandpatch. For both species, individuals in the medium size class dominated the population at all sites. Large *Heliocidaris* were rare, but large *Centrostephanus* were quite abundant, particularly at some locations (e.g. Gunshot Figure 2.4.4). Small (<50 mm) individuals were not abundant, but it is not clear to what extent this was caused by inefficient sampling of this size class.

There was significant variation in the proportion of individuals with roe of a Good colour for both *Centrostephanus* and *Heliocidaris* (Figure 2.4.5, Table 2.4.3). For *Centrostephanus*, a significantly higher proportion of individuals at Little Ramhead had roe of a Good colour than in other locations (SNK, $P < 0.05$). For *Heliocidaris*, a significantly higher proportion of individuals at Bastion Point had roe of a Good colour than in other locations (SNK, $P < 0.05$). In general, few *Heliocidaris* had roe of a Good colour compared to *Centrostephanus* (Figure 2.4.5). For example, the maximum proportion of *Heliocidaris* with roe of a Good colour was 0.2 at Bastion Point, in contrast to 0.7 for *Centrostephanus* at Little Ramhead (Figure 2.4.5).

The area of habitat within each location varied from 3 ha at Tullburnga Island to 951 ha at Airport. The estimated total biomass of *Centrostephanus* within eastern Victoria was about 3300 t, with the greatest biomass occurring at Airport, Gunshot and Sandpatch (Table 2.4.4). The estimated total biomass of *Heliocidaris* within eastern Victoria was about 1500 t. A significant proportion of both of these populations were comprised of individuals with poor quality roe (Figure 2.4.5).

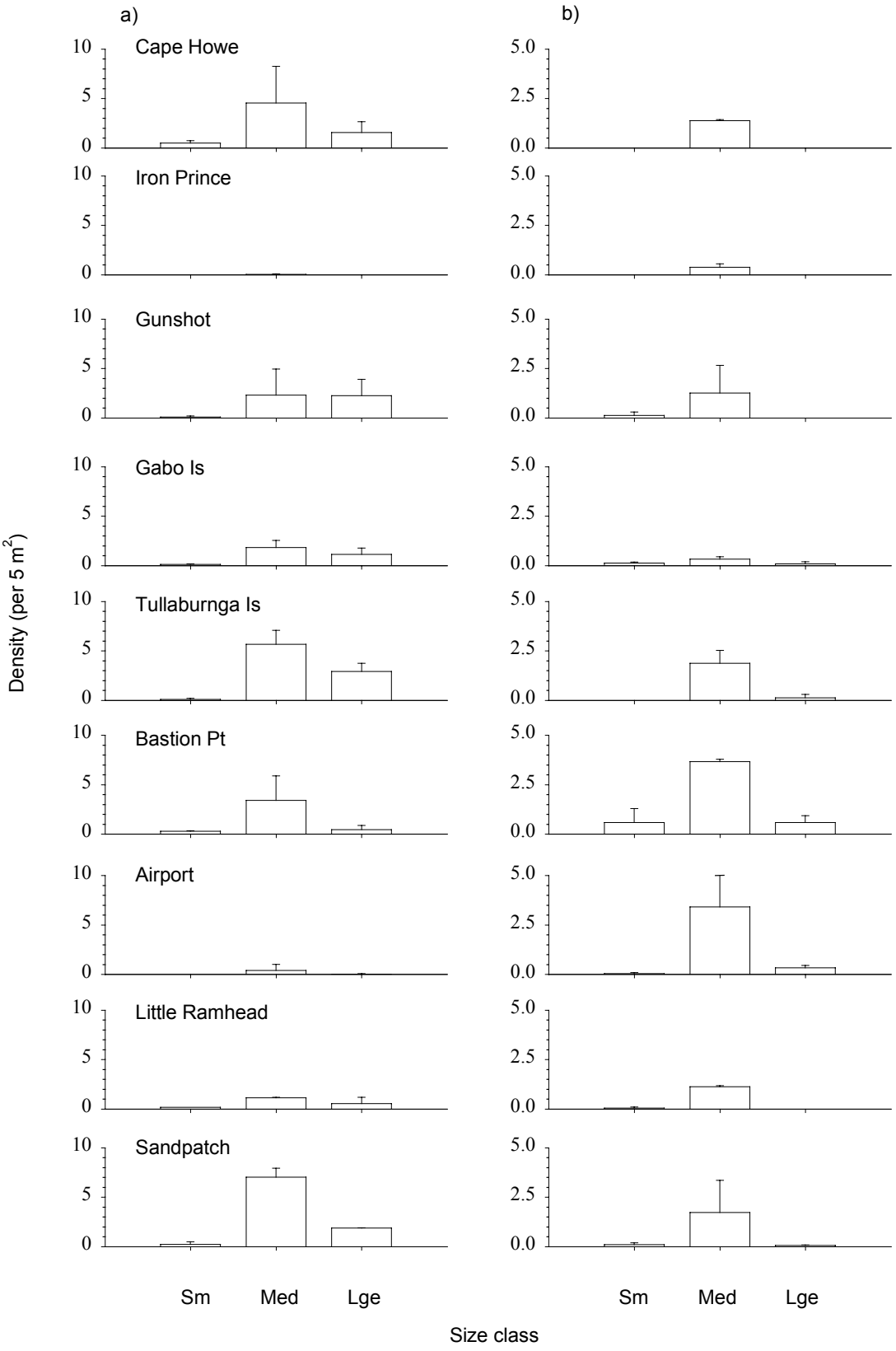

Figure 2.4.4. Density (+ SE) of small medium and large individuals of a) *Centrostephanus* and b) *H. erythrogramma* at nine locations during 2000.

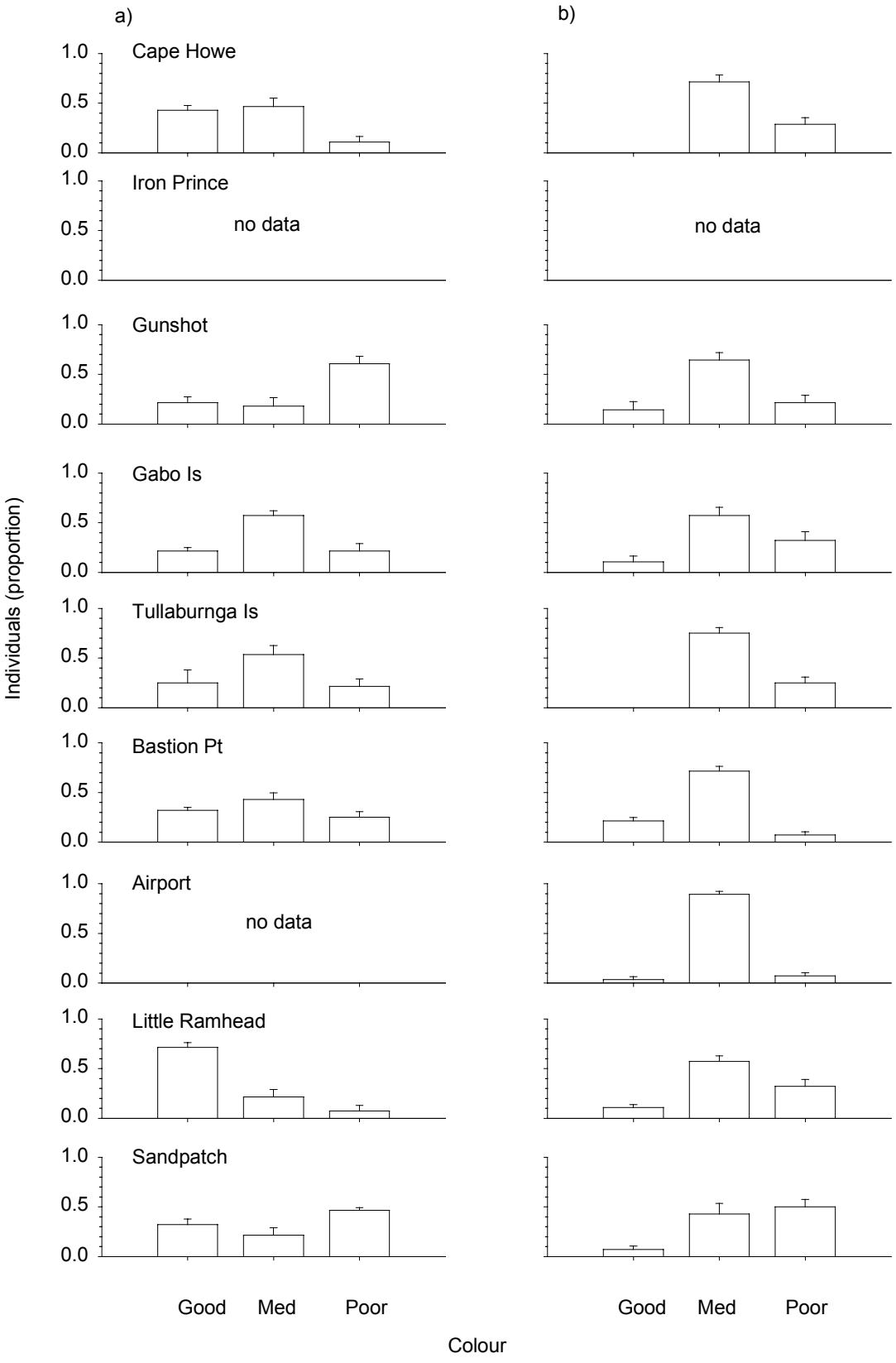

Figure 2.4.5. Proportion (+ SE) of individuals with roe of Good, Medium and Poor colour for a) Centrostephanus and b) *H. erythrogramma* at nine locations during 2000.

Table 2.4.3. Summary of analysis of variance in the density of small, medium and large individuals, and the proportion of individuals with roe of a marketable colour of a) *Centrostephanus*, and b) *H. erythrogramma* in eastern Victoria. An * shows significant effects (P <0.05), [a] = arcsine transformation, and [l] = log transformation.

a) *Centrostephanus*

	Small [l]				Medium				Large			
Source	**df**	**MS**	**F**	**% Var**	**df**	**MS**	**F**	**% Var**	**df**	**MS**	**F**	**% Var**
Location, L	8	0.12	1.66	2	8	184.98	4.41 *	21	8	26.58	3.51 *	17
Tape, Ta(L)	9	0.07	0.86	0	9	41.95	2.10 *	7	9	7.56	2.08 *	7
Res	222	0.08		98	222	19.97		72	222	3.63		76
Total	239				239				239			

	Colour [a]			
Source	**df**	**MS**	**F**	**% Var**
Location, L	6	0.15	4.08 *	49
Res	21	0.03		51
Total	27			

b) *Heliocidaris*

	Small [l]				Medium				Large [l]			
Source	**df**	**MS**	**F**	**% Var**	**df**	**MS**	**F**	**% Var**	**df**	**MS**	**F**	**% Var**
Location, L	8	0.17	1.12	1	8	33.32	1.96	8	8	0.22	5.05 *	9
Tape, Ta(L)	9	0.15	3.14 *	16	9	17.00	2.29 *	9	9	0.04	0.64	0
Res	222	0.04		83	222	7.43		83	222	0.06		91
Total	239				239				239			

	Colour [a]			
Source	**df**	**MS**	**F**	**% Var**
Location, L	7	0.10	2.43 *	22
Res	24	0.00		78
Total	31			

Table 2.4.4. Estimates of the density and size of individuals, and reef area with the calculated biomass (with SE) of a) *Centrostephanus*, and b) *H. erythrogramma* from nine locations in eastern Victoria.

a) *Centrostephanus*

		Cape Howe	Iron Prince	Gunshot	Gabo Island	Tullaburnga Island	Bastion Point	Airport	Little Ramhead	Sandpatch
Density (m^{-2})	A	1.32 (0.65)	0.01 (0.01)	0.93 (0.63)	0.62 (0.20)	1.73 (0.33)	0.83 (0.28)	0.09 (0.09)	0.37 (0.08)	1.83 (0.10)
Weight (kg)	B	0.50 (0.02)	0.49 (0.03)	0.49 (0.03)	0.47 (0.02)	0.39 (0.03)	0.32 (0.02)	0.32 (0.02)	0.46 (0.02)	0.44 (0.02)
Reef area (ha)	C	33	99	232	116	3	11	951	128	142
Biomass (t)	AxBxC	218 (108)	4 (4)	1057 (717)	334 (110)	24 (5)	30 (10)	278 (278)	216 (50)	1146 (78)

b) *Heliocidaris*

		Cape Howe	Iron Prince	Gunshot	Gabo Island	Tullaburnga Island	Bastion Point	Airport	Little Ramhead	Sandpatch
Density (m^{-2})	A	0.28 (0.01)	0.08 (0.03)	0.28 (0.23)	0.11 (0.01)	0.40 (0.12)	0.97 (0.17)	0.76 (0.34)	0.23 (0.01)	0.38 (0.25)
Weight (kg)	B	0.12 (0.01)	0.18 (0.01)	0.18 (0.01)	0.14 (0.01)	0.13 (0.01)	0.25 (0.01)	0.17 (0.01)	0.15 (0.01)	0.16 (0.01)
Reef area (ha)	C	33	99	232	116	3	11	951	128	142
Biomass (t)	AxBxC	11 (1)	13 (4)	112 (92)	18 (2)	2 (1)	27 (5)	1202 (548)	44 (2)	84 (55)

2.4.4. Discussion

Heliocidaris is abundant in Port Phillip Bay wherever solid reef occurs, although densities vary at a range of spatial scales. Densities on the western side of the bay were generally greater than those on the east, although the proportion of individuals with Good quality roe was higher on the eastern side. To some extent, the decreased frequency of Good quality roe on the western side of the bay may have been caused by the greater catches there in recent years. Differences in the yield and colour of roe among locations were also related. That is, in locations where yield was Good, the colour of the roe was also Good. Co-variation in yield and colour of roe could have important implications for patterns of exploitation in the fishery. For example, the fishery will initially target locations with a high proportion of individuals with good yield and colour (e.g. Point Cook). Because of the limited ability of *Heliocidaris* larvae to disperse, depleted populations are likely to receive little recruitment from other locations. Further, only low densities of small individuals were observed, suggesting low recruitment, although variation in recruitment among years is common in sea urchin populations (e.g. Barker 2001). Regardless, these factors will combine to make the population vulnerable to the serial depletion of stocks. As a result, any TAC for the fishery may need to be allocated in proportion to the relative size of the biomass in different areas.

Estimates of the biomass of *Heliocidaris* in Port Phillip Bay can be combined with estimates of the likely productivity of the population to calculate potentially sustainable catches. Likely productivity can be estimated from rates of growth and mortality together with an assumed relationship between stock and recruitment (Francis 1993). Deterministic estimates of MSY for *Heliocidaris* suggest annual catches of 1-5% of the unexploited biomass (Blount and Worthington, unpublished data). The estimated biomass of *Heliocidaris* on reef with macro-algae (i.e. the preferred habitat for harvest because of the yield of roe) of about 4800 t may be close to the unexploited level because of limited fishing to date. Excluding Portarlington (i.e. because of uncertainty over changes in the area of reef), the biomass of *Heliocidaris* on reef with macro-algae was about 3000 t, and suggests catches of 30-150 t could be sustainable independent of the population on bare reef. Unfortunately, some proportion (i.e. ~15%) of this biomass will contain individuals with roe of a quality that is not marketable. As discussed above, catches should also be spread among locations in proportion to the available biomass to avoid serial depletion. Further, because of uncertainties about the potential productivity of the population, conservative catches should be taken until more is known about how the stock responds to depletion. There was also a large biomass of *Heliocidaris* estimated to occur on bare reef (i.e. 4300 t). While a higher proportion of these individuals had roe of a quality that may not be marketable (i.e. ~45%), this population may still contribute to reproductive success within the bay. Although presently not harvested, there is potential to increase catches from habitats that are currently bare reef following the use of techniques, such as those developed for *Centrostephanus*, to improve the quality of roe (see Chapters 3.2 and 3.3). Regardless, if the fishery for these species continues to develop, regular monitoring of the population should continue.

In eastern Victoria, *Centrostephanus* and *Heliocidaris* were present at all locations sampled between Cape Howe and Sandpatch Point. However, *Heliocidaris* was generally much less abundant than *Centrostephanus*. Densities of both species were much less than *Heliocidaris* in Port Phillip Bay, and were more comparable to densities of *Centrostephanus* within Fringe in NSW (see Chapters 2.2 and 2.3). Densities of both species varied significantly at several spatial scales. Despite that, a large proportion of the variation in abundance occurred at the smallest scale sampled, among transects. This is also similar to what occurred for *Centrostephanus* and *Heliocidaris tuberculata* in NSW. Densities of small individuals of both species in eastern Victoria were low and were also comparable to densities of small sea urchins in NSW. Again, this suggests low levels of recruitment, although variation in recruitment among years is common in sea urchin populations. Patterns of variation in colour of roe of *Centrostephanus* and *Heliocidaris* in eastern

Victoria were similar. The proportion of individuals with roe of a Good colour varied between locations for both species. Despite that, few locations had many individuals of either species with roe of Good colour. Most individuals of both species had roe with Medium colour.

Estimates of the biomass of *Centrostephanus* and *Heliocidaris* in eastern Victoria can be combined with estimates of the likely productivity of the population to calculate potentially sustainable catches. Likely productivity can be estimated from rates of growth and mortality together with an assumed relationship between stock and recruitment (Francis 1993). Deterministic estimates of MSY for *Centrostephanus* suggest annual catches of 1-5% of the unexploited biomass (Blount and Worthington, unpublished data). The estimated biomass of *Centrostephanus* of about 3300 t and *Heliocidaris* of about 1500 t may be close to the unexploited level because of limited fishing to date, and suggests catches of 30-150 t of *Centrostephanus* and 15-75 t of *Heliocidaris* may be sustainable. A significant proportion (i.e. ~25%) of this biomass will contain individuals with roe of a quality that is not marketable. Further, these estimates also assume that the populations in eastern Victoria are independent of those in NSW, which is almost certainly not the case, particularly for *Centrostephanus* which is able to disperse larvae to a greater degree than *Heliocidaris*. Larger catches may be sustainable from the population in eastern Victoria if significant recruitment came from less depleted stocks in NSW. Similarly, there may be significant populations of *Heliocidaris* south and west of Sandpatch that could contribute to recruitment in the surveyed area. Similar to the fisheries for sea urchins in NSW and Port Phillip Bay, catches will need to be spread along the coast in proportion to the available biomass to avoid serial depletion. Further, because of uncertainties about the potential productivity of the population, conservative catches should be taken until more is known about how the stock responds to depletion. For *Centrostephanus*, some information is already available as to how the stock may respond to reductions in density resulting from fishing. For example, Blount *et al.* (Chapter 3.2) have shown that recruitment of *Centrostephanus* may increase when densities are reduced, in contrast to what is known for some other species of sea urchin where populations show depensatory responses to harvesting (Tegner and Dayton 1977). In addition, Blount *et al*. (Chapters 3.3, 3.4) has shown the potential of techniques to enhance the quality of roe in *Centrostephanus*. Regardless, if the fishery for these species continues to develop, regular monitoring of the population should continue.

3. DEVELOPMENT OF METHODS TO ENHANCE THE ROE OF SEA URCHINS

3.1. General introduction

The value of sea urchin roe in markets is related to many factors. Size and colour of the roe appear particularly important (Kato and Schroeter 1985). Many studies have attempted to enhance the size and colour of roe by increasing the availability of natural or artificial foods to individuals in aquaria (e.g. Klinger *et al.* 1997, Goebel and Barker 1998). Similar studies that have attempted to enhance the roe of wild populations of sea urchins are rare. Further, it is common for many individuals in populations of sea urchins to have roe that is not of a marketable size or quality (Blount and Worthington 2002). This can result in waste, because a significant proportion of harvested animals contain unmarketable roe and may be discarded. Variability in the quality of roe can also cause fishing effort to be concentrated on areas where it is known a high proportion of individuals have roe of a marketable size or quality. These problems appear to complicate the management of many fisheries for sea urchins throughout the world (Andrew *et al.* 2002).

Among other factors, variability in the quality of roe among individuals is thought to occur because of variation in the availability of food. Indeed, there are many studies of sea urchins kept in aquaria that have shown size and quality of roe are strongly effected by the amount and quality of food available (e.g. de Jong-Westman 1995). This has been taken further to imply that competition among individuals in the wild, for the limited resource of food, is a major cause of variation in the quality of roe (Meidel and Scheibling 1998). This suggests two ways to improve access to food and potentially enhance the quality of roe in the wild. First, the density of individuals can be reduced where it is high, to increase the availability of food to those that remain. Second, individuals can be transplanted to areas where food is more available (e.g. Barrens to Fringe). Experiments investigating changes in density (i.e. through thinning or transplanting) can also provide information about the potential changes in the productivity of a population following changes in density (i.e. depletion from fishing). That is, any compensatory or depensatory responses to changes in density. Further, they can also allow an assessment of likely effects of reducing sea urchin densities on other species of algae and invertebrates.

Changes in the density of individuals can influence a variety of biological and ecological processes that, in turn, can effect rates of growth, mortality and reproduction (Branch and Branch, 1980; Fletcher 1984, see Underwood 1979, Andrew 1989 for reviews). Where species are harvested, this can be used to increase the productivity of a population, as is done in many terrestrial ecosystems (e.g. Thomas *et al.* 1999). However, unless changes in density are done in a controlled manner, reductions in productivity to the target species, and indirect effects on other species can also occur. Unfortunately, for many harvested species, evidence showing that reductions in density can enhance growth and reproduction only becomes apparent after fishing has caused large declines to the population (Gwyther and McShane 1988, Rose *et al*. 2001). The limited development of the fisheries for sea urchins in NSW and eastern Victoria provide the opportunity to complete manipulative experiments investigating the effects of reduced density and use the knowledge gained during management of any development of the fisheries.

3.2. Enhancing the roe of the purple sea urchin by reducing density

C. Blount, D.G. Worthington, K. Organ and R.C. Chick

3.2.1. Introduction

Sea urchins provide an ideal opportunity for investigating models concerning the effects of changes in local density on biological and ecological processes. It is common to find individuals living at a wide range of densities, and their relatively sedentary habit makes sea urchins amenable to manipulative experiments (see Lawrence 1975, Lang and Mann 1976, Lawrence and Sammarco 1982 for reviews). When at high densities sea urchins in many areas of the world can create or maintain areas dominated by a very high cover of crustose coralline algae, that are often referred to as Barrens (e.g. Underwood *et al.*, 1991). When sea urchins are removed from these areas increases in the cover of foliose algae have frequently been recorded (e.g. Duggins 1980, Himmelman *et al.* 1983, Fletcher 1987). Sea urchins are able to rapidly exploit increased resources, and frequently allocate any increased resources to roe production (e.g. Levitan 1991, Russell 1998, Guillou and Lumingas 1999, Guillou *et al.* 2000).

In NSW, fishers of the sea urchin *Centrostephanus rodgersii* avoid catching individuals in Barrens, where the size and quality of roe is poor for the market, and target individuals in Fringe. Fringe is characterised by an abundance of foliose algae (Underwood *et al.* 1991), and the roe of sea urchins is generally larger and of a colour more preferred by the market. Regardless, Barrens can represent >50% of the reef less than 150 m from shore, and contains dense aggregations of sea urchins (Andrew and O'Neill 2000). As a result, a large proportion of the population exists in Barrens (see Chapter 2.3) and this represents a large, un-exploited resource that could be used to reduce pressure on the population in Fringe. Reducing the local density of sea urchins in Barrens is already used on a small-scale by industry to enhance the roe of sea urchins and the recruitment of abalone (Andrew *et al.* 1998). Further, this practice is likely to become more common as the fishery develops. As a result, information about its potential and impacts is needed to facilitate management.

Here we complete experiments at three spatial scales to investigate the effects of reducing the density of *Centrostephanus* in Barrens on the density on the yield and colour of roe. In the small-scale experiment we investigate the effects of density reductions on roe and whether changes occur over a short time period. In the medium-scale experiment we also investigate the effects of density reductions on benthic algal assemblages, and their relationship to changes in the roe and growth of the individuals that remain and the recruitment of juveniles. This experiment repeats the treatments used by Andrew and Underwood (1993), that only investigated the effects of reductions in the density of *Centrostephanus* on the benthic algal assemblage. At the largest-scale we investigate the cost-effectiveness of density reductions in co-operation with industry.

3.2.2. Methods

3.2.2.1. Experimental design and sites sampled

Small-scale

An experiment was designed to investigate the effects of reducing the density of sea urchins on the roe of individuals that remain over a short period of time. To do this, sea urchins were kept in small cages in Barrens. There were three replicate cages for each of four treatments of density. These treatments were repeated over 6 and 3 month periods, starting in December 2000 and March 2001, respectively. The cages had a basal area of 1.25 m^2 and were bolted to the reef. The walls of cages were 10 cm high, 70 cm long, and made of Weldmesh (0.4 mm gauge, 60 mm gap diameter).

The tops were covered with fine chicken wire. Of the four treatments of Density, two were controls (note, additional controls were completed in experiments investigating transplants and were not repeated here). These treatments were Uncaged (i.e. undisturbed sea urchins without a cage), and Handled and Caged (i.e. a 0% density reduction, where sea urchins were handled and caged at a density equal to the natural density of sea urchins at the site, ~4 m^{-2}). The two other treatments involved caging sea urchins with density reduced by 33% and 66%. This corresponded to absolute densities of 6 and 3 sea urchins per cage. Sea urchins used in the experiment had a test diameter of 75-85 mm.

Medium-scale
An experiment was designed to investigate the effects of reducing the density of sea urchins on the roe of individuals that remained, and the benthic assemblage. Treatments consisted of reductions in the density of sea urchins of 100%, 66%, 33% and 0% (i.e. a control treatment where densities were not reduced). Twelve sites were chosen from areas of Barrens in depths of 5 to 8 m along the coastline of Sydney, and three of these were randomly assigned to each treatment. Sites were separated by between 200 m and 5 km and covered an area of reef of 400 to 600 m^2. Borders to each site were chosen to coincide with changes in habitat, to restrict the movement of sea urchins. Densities of sea urchins prior to manipulation were estimated by counting individuals in ten, haphazardly placed, 10 x 1 m transects within each site. The average density of sea urchins at each site ranged from 3.3 to 4.4 m^{-2}, and there was no significant difference among treatments. In March 1999, the density of sea urchins at each site was reduced to the required levels by removing haphazardly chosen individuals. Sites were revisited approximately every 2 months during the first 12 months of the experiment, and then every 3 months until March 2001, and this maintained the treatments within $\pm$ 20% of the required density of individuals m^{-2}.

Large-scale
An experiment was designed to investigate the cost-effectiveness of reducing the density of sea urchins in Barrens over a large spatial scale. Treatments consisted of reductions in the density of sea urchins of 50% and 0% (i.e. a control treatment where densities were not reduced). Within two locations (i.e. Lennards Is and Long Point) to the north and south of Eden, the treatments were replicated at two sites, each covering an area of reef of 2500 to 5000 m^2. The density reductions were commenced in cooperation with commercial sea urchin divers in December 1999, and revisited haphazardly until March 2002 to maintain the appropriate density.

3.2.2.2. Effects of density on sea urchins

Small-scale
To assess the effects of the treatments on the roe of sea urchins, samples were collected during the main harvesting period. In late May 2001, the total weight of each sea urchin and the weight and colour of its roe were sampled. The colour of the roe was matched against a standard colour chart and grouped into two categories for analysis. The categories were those suitable for different markets (i.e. marketable) and those that could not be marketed (i.e. unmarketable), and were determined in cooperation with industry. Marketable roe consisted of a range of colours from bright yellow to orange, whilst unmarketable roe ranged from dark orange to brown. The yield of roe from each sea urchin was calculated as a proportion of the total weight of the individual, and averaged across all individuals within the cage. Average yield and the proportion of individuals with roe of a marketable colour were compared using analysis of variance in which Density and Time were considered fixed factors. Homogeneity of variances was assessed using Cochran's tests and data were transformed where appropriate.

Medium-scale
To assess the effects of the treatments on the roe of sea urchins, samples were collected during the main harvesting period. This was done prior to the manipulation of density in March 1999, and

again after the manipulation in March 2000. Following pilot sampling, 3 groups of 7 sea urchins (70-85 mm test diameter) were taken at each sampling time for all treatments except those where 100% of sea urchins had been removed as few remained. Samples were also collected from 3 sites in Fringe during March 2000. The total weight of each sea urchin and the weight of a single roe element were measured, and the colour of roe was matched to a standard colour chart, and then grouped for analysis (i.e. marketable or unmarketable). The yield of roe for each sea urchin was estimated by the weight of roe as a proportion of total weight, as linear regressions found no significant difference in elevation among sites. The proportion of sea urchins within each category of colour was calculated for each sample of 7 sea urchins and compared using analysis of variance. The three factors were Time (i.e. before and after manipulation) and Treatment (i.e. density reduction), which were considered fixed, and the random factor Site which was nested within Treatment. Data were transformed where appropriate, or according to Cochran's test.

To assess the effects of the treatments on the fecundity of sea urchins, samples were collected immediately prior to the main spawning period in May 2000 (King *et al.* 1994). Pilot sampling suggested there was no difference in the diameter or density of eggs between the interior and exterior of the roe in three sections (i.e. aboral, middle and oral). From this information, a design was chosen that involved removing a single roe element from 8 sea urchins, which were placed in fixative (i.e. 10% formalin), and the removal of 4 samples of eggs weighing between 100 and 500 mg from haphazardly chosen areas of the roe. Estimates of the number of eggs per sample and the weight of each sample were then used with the total weight of the roe to provide an estimate of fecundity for each individual. Differences in fecundity were investigated using analysis of variance at two scales. The first analysis investigated variation in the fecundity of individual sea urchins, where the factor Site was nested within Treatment. The second analysis investigated variation in the fecundity of sea urchins m^{-2}, which was calculated using average estimates of fecundity of individuals and the density of individuals m^{-2} for each site. This analysis involved the single fixed factor of Treatment.

To assess the effects of the treatments on the recruitment of juvenile sea urchins, samples were collected 2 years after the manipulations, in March 2001. This involved counting the number of individuals in two size classes (<30 mm and 30-50 mm test diameter) within ten, 10 x 1 m transects at each site. Based on estimates of growth (see Andrew 1991) individuals <30 mm were likely to be less than one year old, and individuals between 30-50 mm likely to have been between one and two years old, and hence all settled and recruited since the manipulations in March 1999. Differences in recruitment were investigated using analysis of variance, where the random factor of Site was nested within the fixed Treatment factor.

To assess the effects of the treatments on the growth of sea urchins, 250 sea urchins from a marked area within each site were each injected with 2 ml of 1% oxy-tetracycline (OTC). Pilot studies suggested this dose produced a reliable mark in calcified structures of the urchin. Tagging was done in June 1999, 3 months after the manipulations of density. Individuals to be tagged were gently removed from the reef by divers, and a needle used to inject OTC through the peristomal membrane into the animal. Individuals were immediately returned to the reef after injection. Two years after the manipulations, in June 2002, 300 sea urchins were sampled from each site in an attempt to recapture tagged individuals. For each individual the test diameter was recorded and the Aristotle's lantern placed in bleach for 24-48 hours to clean the half-demipyramids of flesh. Once dry, a single half-demipyramid (hereafter referred to as jaw) from each individual was mounted and viewed ventrally under a compound microscope (20×), and exposed to a beam of ultra-violet light, to check for the presence of an OTC mark. Individuals from sites were pooled for each treatment, giving 50 recaptures in the 0% treatment, 72 in the 33% treatment, and 43 in the 66% treatment. Using image analysis, measurements were made of the distance from the origin of each jaw to the OTC mark, and the distance from the OTC mark to the growing margin of the jaw. Growth rates of sea urchins in treatments were estimated using the techniques described in Francis (1995). This

includes estimating the average annual growth rates (i.e. g_1 and g_2) at two sizes (i.e. 15 mm and 22 mm), and the general shape of the growth curve (i.e. b). Other parameters included in the model describe random errors in measurement (i.e. μ_m = 0.022 calculated from repeat measurements), and a parameter describing variation in growth (i.e. v where $\sigma_g = v\mu_g$.).

Large-scale
In March 2002, 6 samples of 7 sea urchins (70-85 mm test diameter) were taken to compare the yield and colour of roe from sea urchins in the different treatments. The total weight of each sea urchin and the weight of a single roe element were measured, and the colour of roe was matched to a standard colour chart, and then grouped (i.e. marketable or unmarketable). The yield of roe for each sea urchin was estimated by the weight of roe as a proportion of total weight. To estimate the changes in roe caused by the density reductions, and the commercial benefits of this technique, two commercial fishers collected between 35-165 kg of sea urchins from each site, and from 3 adjacent sites in Fringe where commercial harvesting of sea urchins usually occurs. The catch was weighed and roe processed for commercial sale. This required processing roe through various washes and draining the end product to produce an estimate of the commercial recovery. The processor also graded the roe of sea urchins from each treatment, and gave an indication of their relative commercial value.

3.2.2.3. Effects of density on benthic assemblages

Results are presented here for the effect of density reductions on the benthic assemblage only for the medium-scale experiment. Samples were collected prior to the manipulations in March 1999, and again in March 2000. The type of substrate or benthic taxa was recorded at 10 equally-spaced (i.e. ~5 cm) points within 5 haphazard-selected areas within 10, 10 × 1 m transects at each site. Where possible, organisms were recorded to the level of species, but others were grouped as follows; crustose coralline algae (*Lithothamnion sp., Neogoniolithon sp., Porolithion sp.* etc), turfing algae (*Amphiroa spp., Corallina spp.*), filamentous algae (*Polysiphonia spp*., *Herposiphonian spp*. etc), *Ralfsia spp*., bryozoans, coral, ascidians, sponges and barnacles. Differences in the cover of the five most common groups of algae (crustose coralline, filamentous, *Ralfsia*, foliose and turfing algae) and the combined cover of sessile invertebrates were investigated using analysis of variance, where the factor Transect was nested within Site, which was nested within Treatment. Treatment was considered a fixed factor whilst Transect and Site were considered random. Data were transformed where appropriate or according to Cochran's test.

Estimates of cover from within each transect were pooled for multi-variate analysis. Bray-Curtis dis-similarity measures were calculated from un-transformed data for each comparison among treatments, both prior to and after the manipulations. These measures were then used in non-metric MDS and compared using analysis of similarities (see Clarke 1993). When comparisons were significant, pairwise tests were used to determine which treatments differed, with the significance level set at 0.10 because of the limited permutations available. The contribution of species or categories that contributed most to the measures of similarity among treatments, and measures of dissimilarity between treatments were also identified (see Clarke 1993). Forward selection regression was used to estimate the relationship between the yield and colour of roe and five of the most common benthic taxa (filamentous algae, *Ralfsia*, foliose algae, turfing algae and invertebrates).

3.2.3. Results

3.2.3.1. Effects of density on sea urchins

In the small-scale experiment, there were significant differences among treatments in the yield of roe (Figure 3.2.1, Table 3.2.1). Yield of roe in treatments where the density had been reduced by 66% were significantly greater than in other treatments (SNK, P <0.05). Where sea urchins were caged for 3 months, yield was increased by 210% from sea urchins in Barrens, and by 169% where sea urchins were caged for 6 months. There was not significant difference between the 3 and 6 month treatments (SNK, P >0.05). There was no significant increase to the proportion of individuals with roe of a marketable colour (Figure 3.2.1, Table 3.2.1).

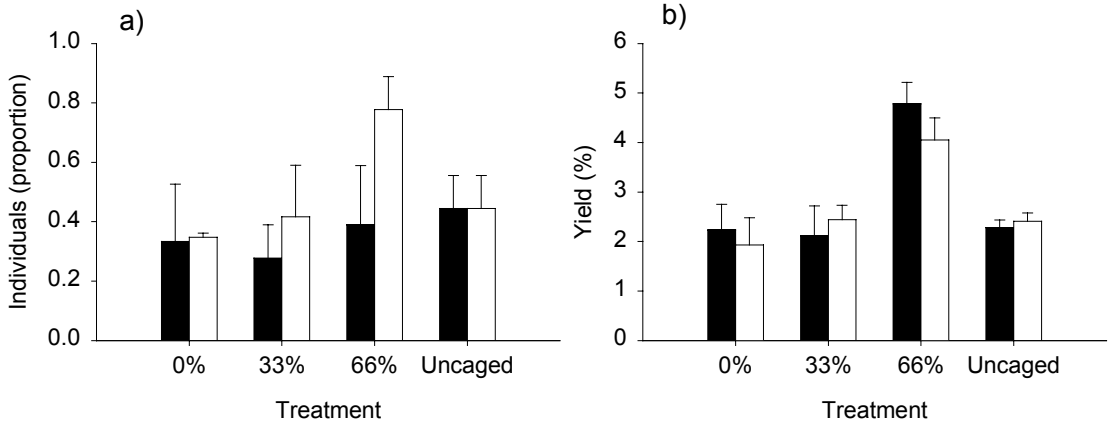

Figure 3.2.1. a) Proportion (+ SE) of individuals with roe of a marketable colour, and b) yield (+ SE) of roe from individuals in different treatments after three months (closed bar) and six months (open bar) in the small-scale experiment.

Table 3.2.1. Summary of analysis of variance in colour and yield of roe of sea urchins caged in Barrens in the small-scale experiment. An * shows significant effects (P <0.05), and [a] = arcsine transformation.

Yield				**Colour** [a]			
Source	**df**	**MS**	**F**	**Source**	**df**	**MS**	**F**
Density D	3	7.23	13.17*	Density D	3	0.11	0.98
Time Ti	1	0.13	0.23*	Time Ti	1	0.26	2.49
D x Ti	3	0.33	0.59	D x Ti	3	0.09	0.92
Res	16	0.55		Res	16	0.11	
Total	23			Total	23		

In the medium-scale experiment, differences in the yield and colour of roe among treatments changed after the experimental manipulation (Table 3.2.2, Figure 3.2.2). In treatments where density was reduced by 66%, both the yield of roe, and the proportion of individuals with roe of a marketable colour, increased to significantly higher levels than the treatment where density was not reduced. There was no significant change in the yield or colour of roe from sea urchins where density was not reduced, or in the colour of roe where density was reduced by 33% (Figure 3.2.2). In all other comparisons among treatments, larger reductions in density significantly increased the yield and colour of roe (Figure 3.2.2). In the treatment that reduced the density of sea urchins by 66%, yield increased 212% and colour by 133% compared to what was found in unmanipulated Barrens. This proportion of individuals with roe of a marketable colour was similar to that in adjacent Fringe at the same time, but yield remained about 2% below that in Fringe (Figure 3.2.2).

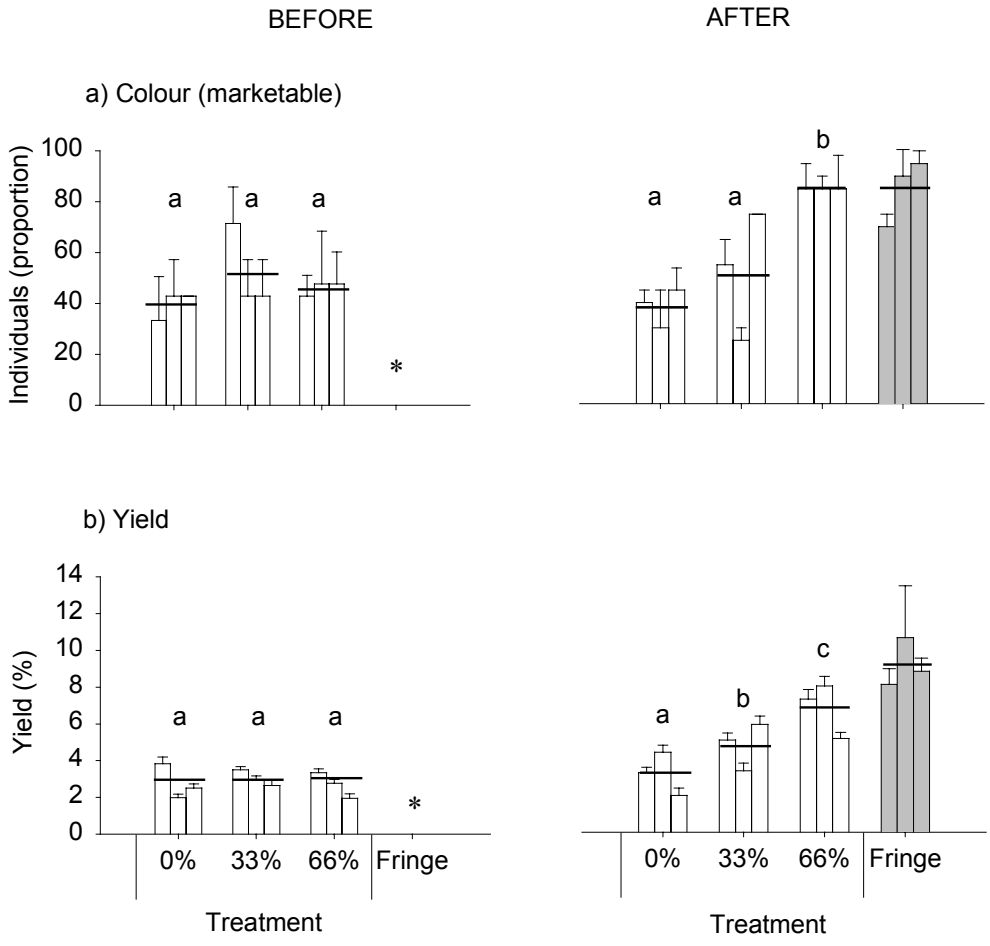

Figure 3.2.2. a) Proportion (+ SE) of individuals with roe of a marketable colour, and b) yield (+ SE) of roe from individuals at three sites within different treatments and adjacent Fringe in the medium-scale experiment. An * denotes data not available, letters show significant differences among treatments, and horizontal lines show treatment means.

Table 3.2.2. Summary of analysis of variance in colour and yield of roe of sea urchins, fecundity of individuals and fecundity per square metre, recruitment, and cover of selected benthic taxa in the medium-scale experiment. An * shows significant effects (P <0.05), [a] = arcsine transformation, and [b] = lognormal transformation.

Roe	Colour [a]					
Source	df	MS	F	df	MS	F
Time, Ti	1	0.29	4.29	1	353.91	17.93 *
Treatment, Tr	2	0.60	8.18 *	2	90.38	3.48
Ti x Tr	2	0.49	7.24 *	2	103.29	5.23 *
Site, Si(Tr)	6	0.07	0.97	6	25.97	11.56 *
Ti x Si(Tr)	6	0.07	0.89	6	19.74	8.79 *
Res	360	0.08		360	2.25	
Total	377			537		

Fecundity x10^{12}	Per individual [b]				Per m^2		
Source	df	MS	F	Source	df	MS	F
Treatment, Tr	2	0.73	2.97	Treatment	2	95.80	76.26 *
Site, Si(Tr)	6	0.25	3.88 *	Res	6	18.60	5.25 *
Res	63	0.06		Total	8		
Total	71						

Recruitment	Juveniles [b]		
Source	df	MS	F
Treatment, Tr	3	9.06	4.82 *
Site, Si(Tr)	8	1.88	4.36 *
Res	108	0.43	
Total	119		

Benthic taxa	Crustose coralline algae [b]			Filamentous algae		*Ralfsia*	
Source	df	MS	F	MS	F	MS	F
Time, Ti	1	3.86	51.64 *	5.58	15.28 *	0.22	4.59 *
Treatment, Tr	3	2.20	4.62 *	1.61	1.83	1.48	3.31
Ti x Tr	3	1.05	14.06 *	1.50	4.10 *	0.24	4.94 *
Site, Si(Tr)	8	0.48	5.51 *	0.88	8.45 *	0.45	10.09 *
Ti x Si(Tr)	8	0.07	1.16	0.36	4.56 *	0.05	0.95
Transplant, Ts(Si(Tr))	108	0.09	1.90 *	0.10	1.94 *	0.04	1.46 *
Ti x Ts(Si(Tr))	108	0.06	1.42 *	0.08	1.50 *	0.05	1.67 *
Res	960	0.04		0.05		0.03	
Total	1199						

	Foliose algae			Turfing algae [b]		Invertebrates	
Source	df	MS	F	MS	F	MS	F
Time, Ti	1	0.12	22.93 *	0.22	3.72	0.17	2.71
Treatment, Tr	3	0.04	9.01 *	0.08	1.41	0.14	0.91
Ti x Tr	3	0.04	1.28 *	0.09	1.63	0.10	1.57
Site, Si(Tr)	8	0.01	6.87 *	0.06	7.52 *	0.15	8.30 *
Ti x Si(Tr)	8	0.01	1.37	0.06	7.41 *	0.06	3.32 *
Transplant, Ts(Si(Tr))	108	0.01	1.05	0.01	1.09	0.02	1.50 *
Ti x Ts(Si(Tr))	108	0.01	1.04	0.01	1.14	0.02	1.51 *
Res	960	0.01		0.01		0.01	
Total	1199						

There was significant variation in the yield of roe among sites after the experimental manipulation. Much of this variation also persisted between the March and May samples, with a significant positive correlation in yield among sites (R^2 = 0.67). Differences in yield among sites and treatments after the experimental manipulation were not reflected in significant differences in fecundity of sea urchins (Figure 3.2.3, Table 3.2.2), despite individuals in the treatment where density was reduced by 66% containing 166% more eggs than where density was not reduced. In contrast, there were significant differences between treatments in the absolute fecundity of sea urchins (Figure 3.2.3, Table 3.2.2). When estimates of the number of eggs per individual were combined with the density of individuals, the treatments that reduced the density of sea urchins contained approximately 48% less eggs m^{-2} than where density was not reduced.

The recruitment of juvenile sea urchins also differed significantly among treatments (Table 3.2.2). Recruitment increased with each treatment that increased the reduction in density of sea urchins. In the treatment where all sea urchins were removed, recruitment was approximately 900% more than that in the treatment where density was not reduced (Figure 3.2.4). There were also more recruits in the second year of the experiment than the first year.

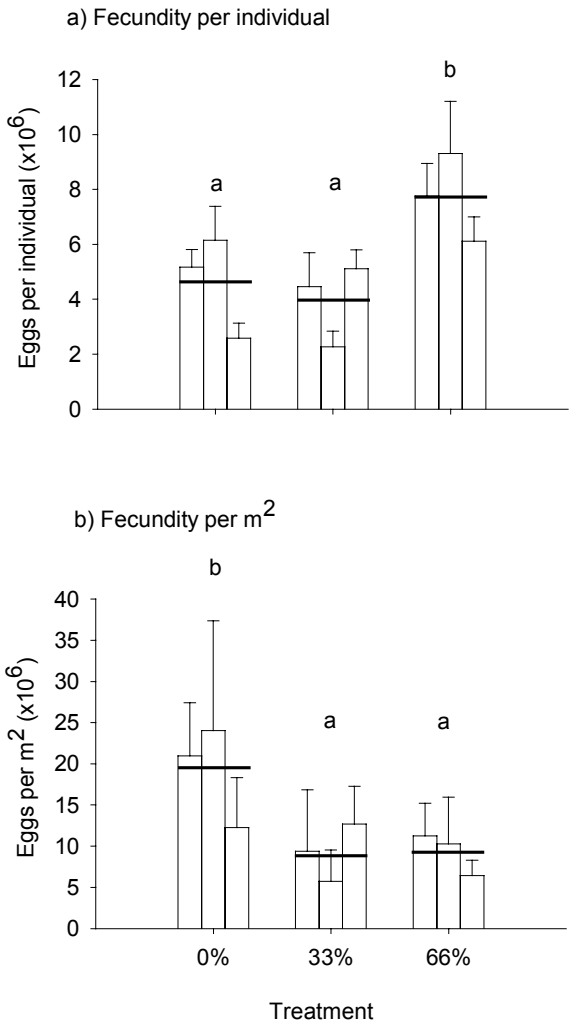

Figure 3.2.3. a) Fecundity (+ SE) of individuals and b) fecundity (+ SE) per m^2 at three sites within different treatments in the medium-scale experiment. Letters show significant differences among treatments, and horizontal lines show treatment means.

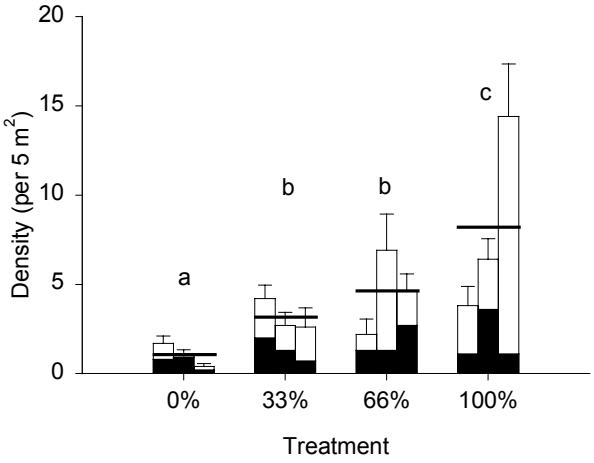

Figure 3.2.4. Density (+ SE) of juveniles at three sites within different treatments in the medium-scale experiment. Open parts of each bar represent the proportion of recruits <30 mm in size (i.e. likely to be 1 year old) and closed bars the proportion of recruits >30 mm and <50 mm in size (i.e. likely to be 2 years old). Letters show significant differences among treatments, and horizontal lines show treatment means.

Average annual growth of the jaw of sea urchins in treatments where the density had been reduced by 33% and 66% was similar, but significantly greater than in the treatment where density remained the same (*t*-test, $P < 0.05$). Differences in the expected annual increment in the jaw within both treatments where density had been reduced were more than double that where density remained the same (Figure 3.2.5a, Table 3.2.3). There was a significant relationship between the length of the jaw and the diameter of the test ($\log_e$ transformed), and this relationship did not vary significantly between treatments (ANCOVA, $P < 0.05$). This relationship was used to estimate the growth of the test (Figure 3.2.5b and c). For example, in treatments where the density had been reduced by 33% and 66%, individuals with a 68 mm test were estimated to take 8 years to reach 90 mm, compared to >25 years in treatments where density was not reduced.

Table 3.2.3. Estimates of growth parameters (with SE) for recaptured sea urchins in treatments in Barrens habitat where 0%, 33% and 66% of individuals had been removed in the medium-scale experiment.

	Treatment		
	0%	**33%**	**66%**
Curve shape			
b	4.619 (0.076)	4. 619 (0.076)	4. 619 (0.076)
Growth rate			
g15	0.653 (0.010)	1.582 (0.014)	1.992 (0.023)
g22	0.204 (0.002)	0.431 (0.002)	0.406 (0.003)
Growth variability			
v	0.562 (0.006)	0.358 (0.003)	0.339 (0.004)
Measurement error			
p	0.022	0.022	0.022

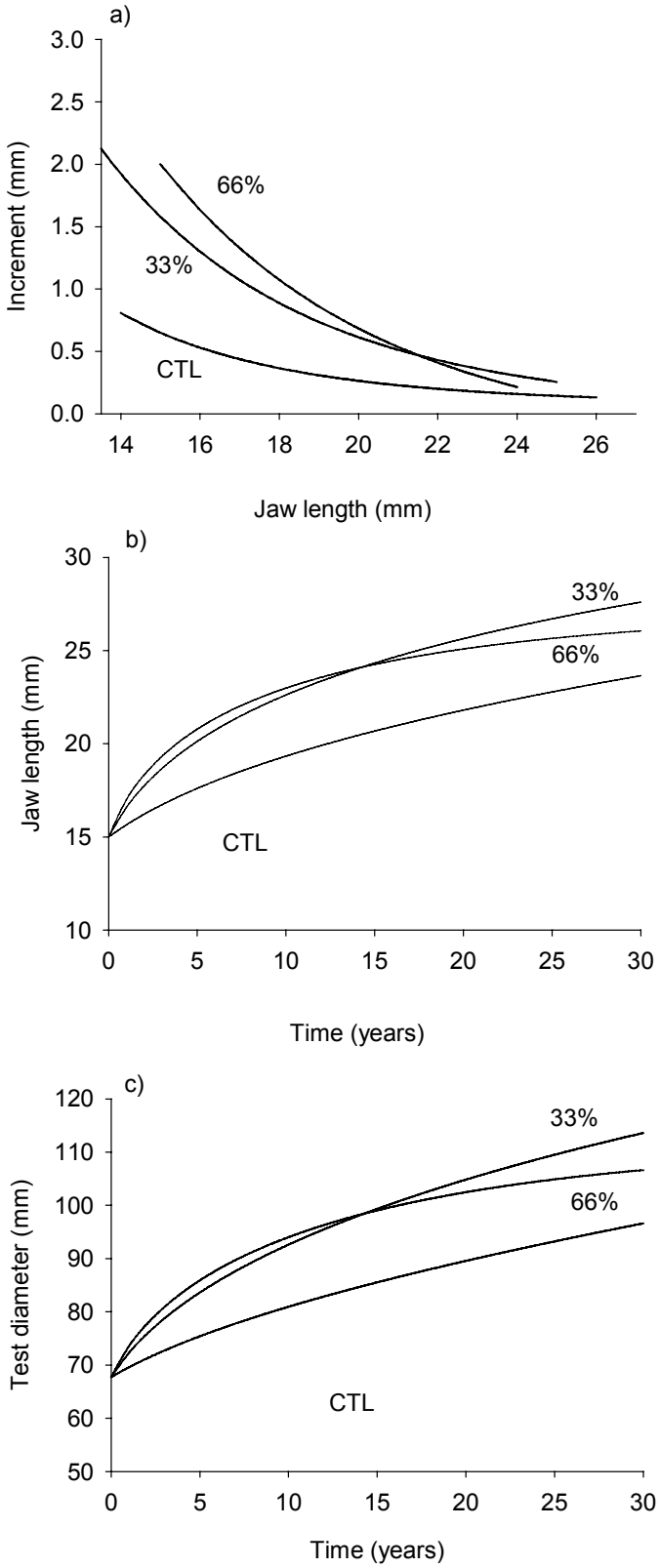

Figure 3.2.5. a) Expected annual growth of the jaw b) expected growth of a 15 mm jaw through time and c) expected growth of a 68 mm test through time within different treatments in the medium-scale experiment. Note, test growth was estimated from growth of the jaw and the relationship between the relative size of the jaw and test.

In the large-scale experiment, there was a significant difference among treatments in the yield of roe after the density reductions. The yield of roe in the treatment where densities had been reduced by 50% was almost double the yield of roe of sea urchins at sites where densities were not reduced (Figure 3.2.6). There was also a significant increase in the proportion of individuals with roe of a marketable colour at Lennards Is, but not at Long Pt (Figure 3.2.6).

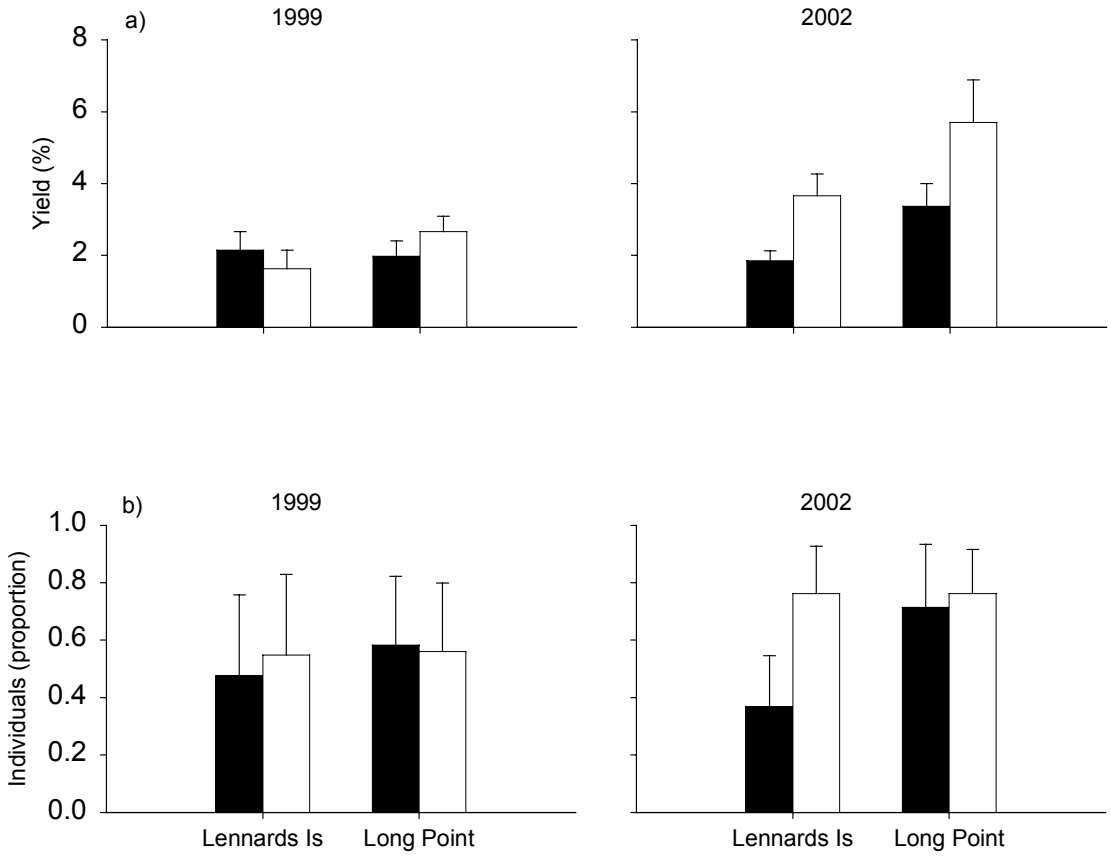

Figure 3.2.6. a) Yield (+ SE) of roe and b) proportion (+ SE) of individuals with roe of marketable colour at two sites before (December 1999) and after (March 2002) density reductions of 0% (closed bars) and 50% (open bars) in the large-scale experiment.

3.2.3.2. Effects of density on benthic assemblages

In the medium-scale experiment, there was significant variation in the benthic assemblage after the experimental manipulation that was not present before (Figure 3.2.7a). Treatments where the density of sea urchins was reduced became more dissimilar to the treatments where density was not reduced (Table 3.2.4). The largest differences were between the treatment where the density was not reduced and the treatments where density was reduced by 66% and 100% (ANOSIM, $R<0.05$). Most (i.e. 90%) of the differences among treatments were related to the abundant taxa of crustose coralline algae, filamentous algae, and *Ralfsia*. Although treatments became less similar after manipulation, sites within treatments were not any more similar than they were before manipulation (Figure 3.2.7b-e), suggesting the assemblage at the site prior to manipulation had a large bearing on the composition of the assemblage observed after manipulation had taken place.

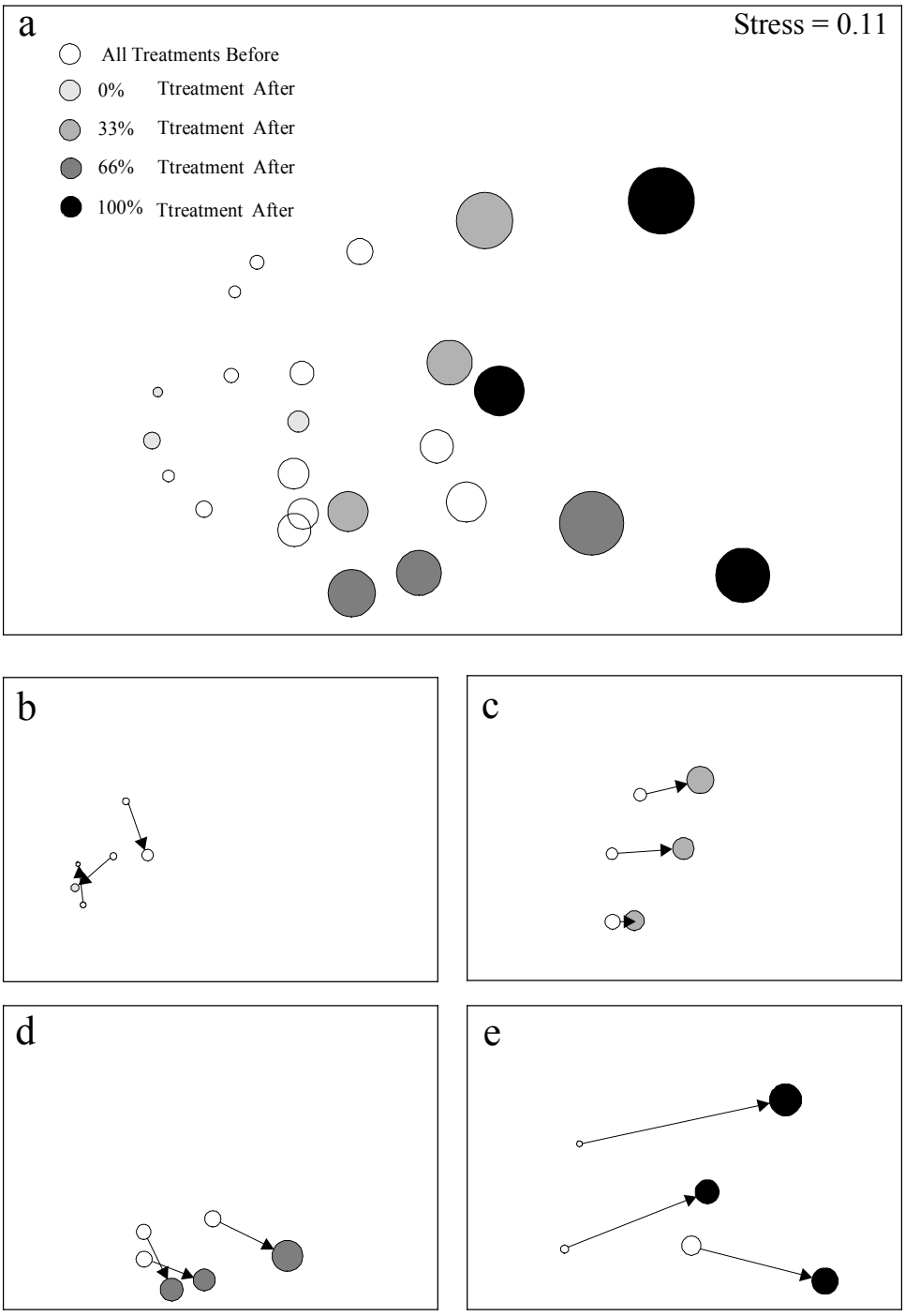

Figure 3.2.7. Two-dimensional nMDS plot of the centroids of benthic assemblages in the medium-scale experiment at a) twelve sites within different treatments before and after density reduction, and b-e) show changes for each site within the 0%, 33%, 66% and 100% treatments, respectively. The size of each circle represents the mean percent cover of filamentous algae and *Ralfsia*, and arrows link sites before and after the density reduction.

Table 3.2.4. Bray-Curtis dissimilarities between each possible pair of treatments before and after manipulations of density in the medium-scale experiment. Values are the mean from three sites.

Bray-Curtis Dissimilarity

		Treatment					
		0%		33%		66%	
		Before	After	Before	After	Before	After
Treatment	33%	25.4	39.1				
	66%	27.5	42.4	24.4	30.9		
	100%	29.9	54.7	28.8	38.2	24.4	40.7

The cover of benthic taxa was dominated by crustose coralline algae, filamentous algae, and *Ralfsia* (i.e. >90%) and although the cover of each increased with the amount of sea urchins removed after one year, relative differences between these types of algae were similar for each treatment. There was a strong association of these particular taxa to differences observed among treatments (Figure 3.2.8). *Ralfsia* and filamentous algae were also closely related to the colour and yield of roe. Much of the variation between sites in colour and yield of roe is related to the cover of these two taxa (Figure 3.2.8; $R^2 = 0.41$ and 0.55, $P < 0.05$).

The cover of the six most common taxa varied significantly among treatments between before and after the experimental manipulation (Table 3.2.2, Figure 3.3.9). Crustose coralline algae were the most common taxa in all treatments before and after manipulations in all cases except the treatment where density was reduced by 100%. After manipulation, the cover of crustose coralline algae in treatments where density had been reduced by 33% and 66% were both reduced to ~25% of their original cover. Where density was reduced by 100%, crustose coralline algae dropped to ~50% of the original cover. Changes in cover of filamentous algae, *Ralfsia* and foliose algae were generally complementary to those for crustose coralline algae (Figure 3.2.9). That is, after manipulations, as the reduction in density increased, the cover of these taxa increased (Figure 3.2.7). Despite that, the cover of *Ralfsia* and foliose algae in treatments where density had been reduced by 100% was less than for the treatments reduced by 66%. There was no significant difference in the cover of turfing algae or sessile invertebrates between before and after the manipulation (Table 3.2.1, Figure 3.2.9). Variation in the cover of these taxa within sites and among sites within treatments was large relative to any differences among treatments.

3.2.3.3. Costs and benefits of reducing density

A total of 183 kg of sea urchin harvested from the large-scale experiment was supplied to the processor by 2 commercial fishers. Before processing, the average yield of roe from sea urchins at sites where density had been reduced was approximately 4.7% of the total weight of individuals, but after processing this dropped to approximately 4.5%. This is about 2% less than yields from commercial catches of sea urchin in Fringe at that time of year. Over 35% of the roe from sea urchins in sites where density had been reduced was considered by the processor to be of high quality. This is also similar to that obtained from commercial catches of sea urchin from Fringe at that time of year. Approximately 25 person days were required to reduce and maintain the reduced density of sea urchins in the large-scale experiment. The harvest of 183 kg at the end of the experiment equated to about 3% of the total amount of sea urchins in the experimental sites. Hence, about 6.5 t of sea urchin remains in the experimental sites, which would be equivalent to

about 12-20 days fishing for a commercial fisher, at current harvesting rates. The processor recovered 8.6 kg of roe from the commercial catch of sea urchins from the large-scale experiment. Combined with the current average market price of \$42 kg^{-1} this equates to a value of \$340. The cost of processing and marketing the roe from the sample of sea urchins taken from the large-scale experiment was estimated by the processor to be approximately \$340, which would have included the cost of buying sea urchins from the commercial fishers (\$130), processing (\$130), and costs associated with transport and miscellaneous (\$80). As a result, the net profit to the processor was only \$20, but it should be noted that this is related to the small catch taken from the experiment.

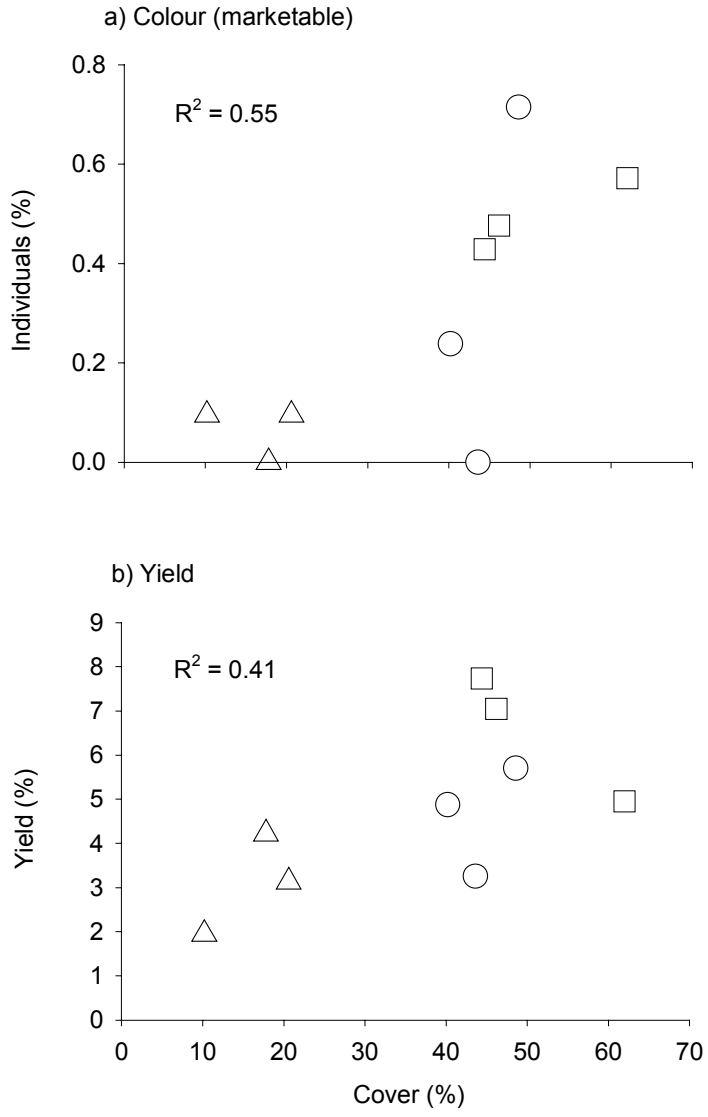

Figure 3.2.8. Relationship between mean percent cover of filamentous algae and *Ralfsia* at sites within different treatments and a) the proportion of individuals with roe of a marketable colour, and b) yield of roe in the medium-scale experiment. Density reduction treatments are shown as 0% = Δ, 33% = O, and 66% = □.

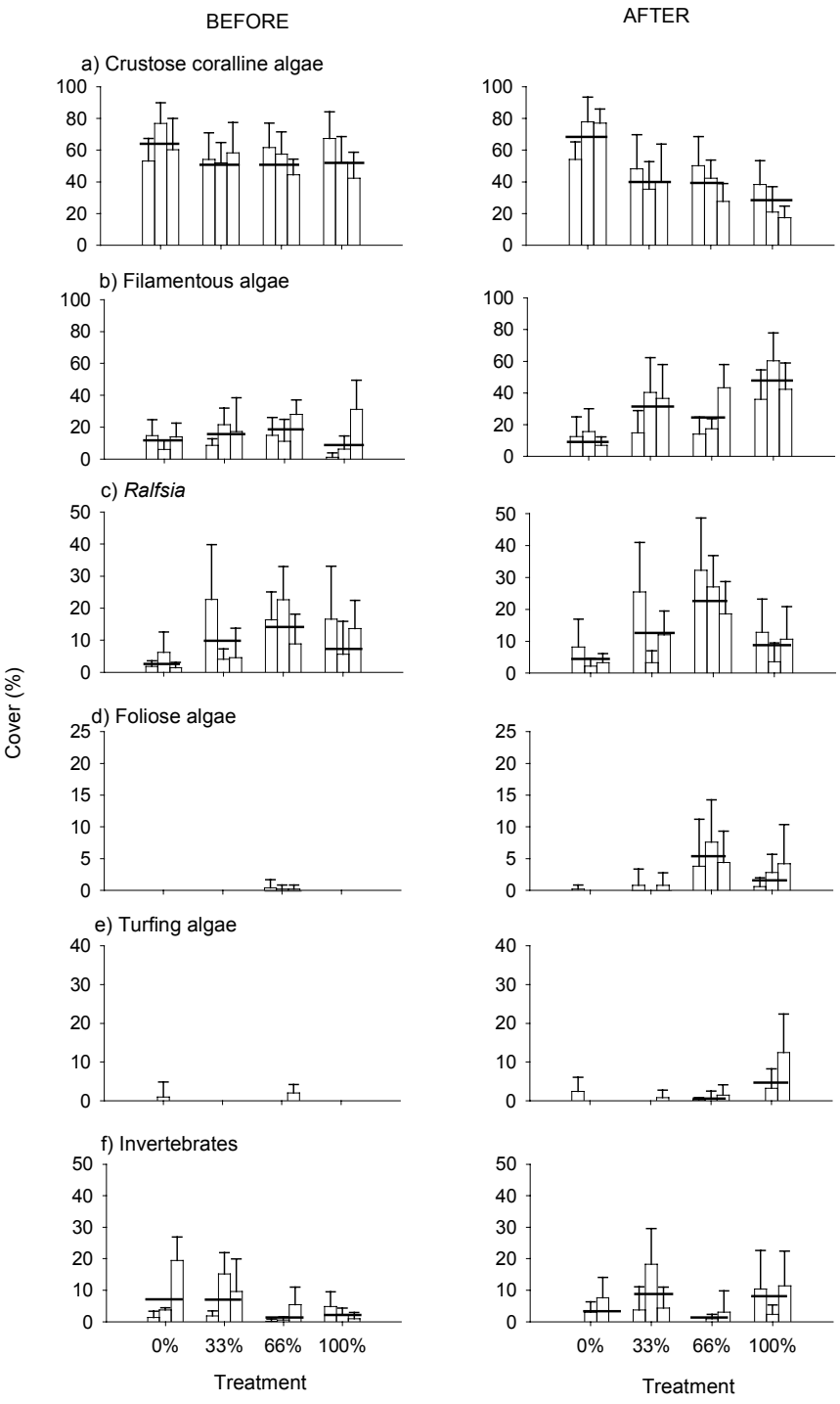

Figure 3.2.9. Percent cover (+ SE) of a) crustose coralline algae, b) filamentous red algae, c) *Ralfsia spp.*, d) foliose algae, e) turfing algae and f) sessile invertebrates at three sites within different treatments in the medium-scale experiment. Horizontal lines show treatment means.

3.2.4. Discussion

Although the roe of sea urchins is related to the supply of food for several species (Vadas 1977, Wahle and Peckham 1999), the potential of using changes in the local density of sea urchins to increase the availability of food and the consequent quality of roe has rarely been investigated. In response to reductions in their local density, we observed significant improvements (i.e. for the market) in the yield and colour of the roe of *Centrostephanus* living in Barrens, and these were observed in relatively short periods of time. Although significant improvements to yield were observed as quickly as 3 months after density reductions were made, larger increases to yield were observed when sea urchins were left for a year after reductions were made. In addition, after one year, significant increases in the yield of roe occurred where density was reduced by as little as 33%, but the greatest increases (i.e. 212%) occurred where 66% of sea urchins were removed. Despite that, yield was rarely enhanced to the level observed for populations in Fringe that are currently preferred by the fishery. Even in the large-scale experiment, which was running for over 2 years, yield of roe from areas where densities had been reduced by 50% were still slightly below those for sea urchins harvested from Fringe. Regardless, yield was improved to a level that industry would consider viable for harvesting, considering some improvement can also be made to colour.

With an abundant population in Fringe there is little incentive to fishers in utilising the much larger population in Barrens. Although yield and colour of roe can be improved by reductions in density, it is costly to reduce and maintain the density of sea urchins. Even at this early stage in the development of the fishery, anecdotal evidence suggests that some productive areas of Fringe have quickly become depleted. It is likely that techniques of density reduction in Barrens will be used more frequently if the fishery were to expand. This could reduce the pressure on sea urchin populations in Fringe, and potentially allow an increase in the available habitat for abalone.

Improvements in the yield and colour of roe may be related to changes in the benthic algal assemblage associated with the reductions in density of *Centrostephanus*. Crustose coralline algae, filamentous algae and *Ralfsia* dominated the benthic assemblage after the density reductions, and variation in the cover of these algae was important in distinguishing between the treatments in the medium-scale experiment. When densities were reduced, the cover of crustose coralline algae decreased, whereas the cover of filamentous algae and *Ralfsia* increased. As a result, the cover of crustose coralline algae was negatively associated with yield and colour of roe, and the cover of filamentous algae and *Ralfsia* was positively associated. Many studies have shown improvements in the yield and colour of roe can be caused by change in the supply of food (eg Russell 1998).

Despite the common relationship between the weight of sea urchin roe and its colour (e.g. Blount and Worthington 2002), simply increasing the yield of roe does not necessarily improve colour (Klinger *et al.* 1997). Carotenoid pigments in the diet are required to produce the yellow or orange colour of roe, and these are usually rapidly converted to the keto carotenoid echinenone (Griffiths and Perrott 1976). Both filamentous algae (*Polysiphonia spp*., *Herposiphonian spp*.), *Ralfsia* and foliose brown algae contain α and β-carotenoids, and fucoxanthin. Increases in the abundance of these filamentous algae, *Ralfsia* and foliose algae in treatments where the density was reduced may have given sea urchins access to carotenoids that were not as common in assemblages of algae prior to removals.

Changes to the demography of *Centrostephanus* were also observed in response to the reductions in density in the medium-scale experiment. The abundance of juvenile *Centrostephanus* was negatively associated with density, suggesting settlement or recruitment was greater at lower densities. This is in contrast to many other species of sea urchin. For example, Tegner and Dayton (1977) showed that juvenile *Strongylocentrotus franciscanus* have lower mortality under the spine

canopy of adults. The increased settlement or recruitment of *Centrostephanus* at lower densities may be related to an increased supply of shelter at settlement or a lower rate of mortality (e.g. interference competition from adults). This compensatory relationship will be beneficial to the productivity of the fishery if it also occurs at a larger spatial scale. Alternatively, reduced densities caused by fishing may initially increase recruitment, but egg production and the supply of larvae may become limiting as depletion increases across a large spatial scale.

The rate of growth of *Centrostephanus* was also faster at lower densities. Similar relationships have also been proposed for other sea urchin species (eg Levitan 1989). Reducing the density of adults by 33% doubled the rate of growth of *Centrostephanus*. Increased growth will cause individuals to reach sizes preferred by the market more rapidly, and they may also mature more rapidly. Rapid growth to larger sizes may also reduce rates of mortality on early life stages, as evidence from other species suggests intermediate-sized individuals are more exposed to predation (Schiebling and Hamm 1991).

While there was no significant difference in the diameter of eggs from sea urchins at different densities, other changes in the roe at lower densities may result in improvements in the quality of the eggs that may effect rates of fertilisation and survival of larvae. For example, reduced gonad size is associated with less viable larvae in some reef fish (McCormick 1998). Despite that, King *et al.* (1994) found little difference in rates of fertilisation for *Centrostephanus* from Fringe and Barrens. Further, rates of successful fertilisation may be more closely related to egg production m^{-2} than rates of individual fecundity. For example, there is evidence suggesting fertilisation rates in sea urchins can decline dramatically when individuals are separated by >1 m (Pennington 1995, Levitan *et al.* 1992). Egg production m^{-2} was lower at reduced densities. More work is needed to understand the dynamics of local density reductions and their effect on egg production, successful fertilisation and larval survival in *Centrostephanus.*

In addition to knowledge about the effects of reductions in density of *Centrostephanus* on the productivity of the population, information is also needed on their impact on other species. Similar information is also collected about the effects of the management other natural resources on associated species (e.g. Thomas *et al.* 1999). For example, while reductions in density are used in forestry to improve productivity, the effects on other species has been well studied. Reductions in density of *Centrostephanus* caused decreases in the cover of crustose coralline algae, and increases in the cover of filamentous and foliose algae in the medium-scale experiment (see also Appendix 7.2). Despite that, the changes in cover of these dominant taxa were not linearly related to the density of sea urchins (see also Andrew and Underwood 1993). That is, the total cover of filamentous and foliose algae was similar following 33% and 66% reductions in density, but increased following 100% reductions. Andrew and Underwood (1993) suggested compensatory changes in the foraging behaviour of individual sea urchins might have caused this non-linearity. The cover of foliose algae in all treatments, and particularly those where density was reduced by 100%, was much less than that reported by Andrew and Underwood (1993). This may be related to the increased area of our sites and the restricted dispersal of some species of algae. Alternatively, our clearances in the medium-scale experiment were started at a different time of year than those made by Andrew and Underwood (1993). Seasonal differences in the recruitment of different algae can cause different successional patterns (Kennelly 1987).

Natural changes in the density of *Centrostephanus* in Barrens appear to be small (Andrew and Underwood 1989). Some large changes in the density of *Centrostephanus* have been observed following floods or storms (Andrew 1991), but these have only taken place on a relatively small spatial scale. Further, there is some evidence suggesting that densities of sea urchins and the area of Barrens may have increased over much of the south coast of NSW in recent decades. These natural changes in the density of *Centrostephanus* need to be considered when assessing the likely impacts of any deliberate density reduction or the effects of the fishery. Regardless, reductions in

the density of *Centrostephanus* offer the potential to increase the productivity of both the sea urchin and abalone populations. Whilst the research presented here provides an initial assessment, more research investigating the effects of reducing sea urchin densities is needed to fully investigate its potential and impacts.

3.3. Enhancing the roe of the purple sea urchin by transplanting individuals

C. Blount, D.G. Worthington and B.R. Stewart

3.3.1. Introduction

Increasing the availability and quality of food can enhance the quality of sea urchin roe (Russell 1998, Guillou and Lumingas 1999, Guillou *et al.* 2000), although this information comes mostly from studies of sea urchins held in aquaria. The supply of food to sea urchins in the wild can be improved in several ways without simply adding the food. First, reducing the density of sea urchins can modify the benthic algal assemblage from one dominated by crustose coralline algae to more filamentous and foliose algae (see Lawrence *et al.* 2001 for most recent review). This has been shown to improve the size and colour of roe for market (see Chapter 3.2). Second, sea urchins can be transplanted to areas where more food is likely to be available. To compare the potential of each of these methods, the relative costs (e.g. in time, money and mortality) and benefits (i.e. money) need to be assessed.

Hatchery-reared sea urchins in Japan are released into areas where food is readily available, and transplanting juvenile sea urchins is being investigated in several other fisheries (see Andrew *et al.* 2002 for review). This can involve the need for large infra-structure and high operating costs. Further, releases involve small individuals that can take several years to reach marketable size. However, associated with many fisheries for sea urchin are some populations that are not targeted by commercial fishers because of the high proportion of individuals with roe of a poor quality. Further, these populations often exist at high densities, allowing little development of the benthic algal assemblage. There is an opportunity to transplant these individuals to areas that are likely to provide a greater availability of food to rapidly enhance their roe.

A small, but expanding, fishery for the sea urchin *Centrostephanus rodgersii* exists in south-eastern Australia. Although the total biomass of *Centrostephanus* is enormous (see Chapter 2.3), most of the population exists in Barrens, where the size and quality of roe is not generally acceptable to the market. *Centrostephanus* is harvested mostly from Fringe (Underwood *et al.* 1991), where foliose algae are more diverse and abundant. The roe of sea urchins in Fringe is generally larger and of a colour more preferred by the market. However, despite the fishery being in an early stage of development, intense local depletion of populations can occur, particularly in areas with roe of good size and quality. Transplanting individuals from Barrens to Fringe is already completed on a small-scale by industry, as it can rapidly enhance the roe, and its use may expand together with the development of the fishery. As a result, information about the potential of transplanting sea urchins is needed to facilitate its management.

In this study we investigate the potential of transplanting *Centrostephanus* from Barrens to Fringe to rapidly enhance the size and colour of their roe. Experiments are completed at three spatial scales to investigate the effects of density and timing of the transplant. Further, at the largest spatial scale we attempt to investigate the cost-effectiveness of transplanting sea urchins. In conjunction with these experiments, we also investigate the effects of the transplants on algal and invertebrate populations (see Appendix 7.2).

3.3.2. Methods

3.3.2.1. Small-scale

A small-scale experiment was designed to investigate the effects of transplanting sea urchins among habitats. Sea urchins were kept within small cages and there were three replicate cages for each of seven treatments. The cages had a basal area of 0.49 m^2 and were bolted to the reef. The walls of cages were 10 cm high and 70 cm long and made of Weldmesh (0.4 mm gauge, 60 mm gap diameter), and the tops were covered with fine chicken wire. Of the seven treatments, four were controls. These treatments were Uncaged (i.e. undisturbed sea urchins without a cage), Caged (i.e. undisturbed sea urchins were caged), Handled (i.e. sea urchins were handled, replaced and caged), and Reduced (i.e. sea urchins were handled and caged with their density reduced by 50%). The three other treatments involved transplanting sea urchins from Barrens to Fringe at a density of 0.5, 1 and 2.5 times the original density in the Barrens (i.e.~4 m^{-2}). This corresponded to absolute densities of 1, 2 and 5 sea urchins per cage. Sea urchins used in the experiment had a test diameter of 75-85 mm.

The experiment began in April 2001 and was sampled after 6 weeks. For each sea urchin, its total weight and the weight of roe were measured, and the colour of the roe was matched against a standard colour chart and grouped into two categories (i.e. marketable or not). The yield of roe from each sea urchin was then calculated as a proportion of the total weight of the individual, and averaged across all individuals within the cage. Average yield and the proportion of individuals with roe of a marketable colour were compared using analysis of variance in which Treatment was considered a fixed factor. Homogeneity of variances was assessed using Cochran's tests and data were transformed where appropriate.

3.3.2.2. Medium-scale

Two sets of treatments were used in an experiment to determine the effects of transplanting sea urchins among habitats. The first treatments involved transplanting sea urchins into cages with several controls. These treatments were used to investigate the relative effects on roe of caging, handling, moving and transplanting sea urchins among habitats. Sea urchins were kept in small cages (i.e. see above) and there were 4 treatments. The treatments were Uncaged (i.e. sea urchins were handled but not caged), Caged (i.e. undisturbed sea urchins were caged), Handled (i.e. sea urchins were handled and replaced), and Moved (i.e. sea urchins were handled, moved and caged). Sea urchins in the Uncaged treatment were relocated at the end of the experiment from underwater maps of the home sites of three pairs of 2 sea urchins living together. There were 2 sea urchins per cage and 2 replicate cages per treatment.

The second set of treatments compared the roe of sea urchins transplanted at different densities into areas of Fringe with those from un-transplanted individuals in Barrens and Fringe. That is, there were five treatments. The treatments were the transplanting of sea urchins into Fringe, that was cleared of its original population, at 0.5, 1 and 2 times the original density, and un-transplanted sea urchins from Barrens and Fringe. There were three, replicate sites within each treatment. Sites ranged in area from ~20-40 m^2, and original populations in each site ranged from 140-220 individuals. In most instances, sites were bounded by natural barriers, making it difficult for transplanted individuals to leave sites, and for other individuals to enter.

Both sets of treatments were set up in December 2001 using sea urchins with a test diameter of 75-85 mm. In March 2002, all sea urchins were collected from the first set of treatments, while 3 samples of 7 sea urchins were collected from each site in the second set. For each sea urchin, the

total weight of the animal and the weight of roe were measured. The yield of roe for each sea urchin was then calculated as a proportion of the total weight. The colour of roe was matched against a standard colour chart grouped into two categories (i.e. marketable or not). Yield and the proportion of individuals with roe of a marketable colour were compared using analysis of variance. In the first set of treatments, the random factor of Cage was nested within the fixed factor Treatment. In the second set of treatments, Treatment was also considered a fixed factor, and, the loss of transplanted sea urchins through the experiment was also calculated. The loss of transplanted sea urchins could have been related to emigration or mortality from handling, or related to changing the density of sea urchins at a site. Loss of sea urchins was compared among treatments, with Treatment considered a fixed factor. Homogeneity of variances was assessed using Cochran's tests and data were transformed where appropriate.

3.3.2.3. Large-scale

An experiment was designed to investigate the cost and benefit of the large-scale transplanting of sea urchins. Two treatments were used at four sites to estimate the benefits of transplanting sea urchins (i.e. improvements to the yield and colour of roe, and mortality of transplanted individuals). The treatments were Transplant (i.e. sea urchins transplanted from Barrens to Fringe) and Control (i.e. sea urchins remained undisturbed in Barrens). Because of the large-scale of the transplant, sea urchins were only transplanted to one site, while three sites were used as controls. The area of reef chosen to receive the transplant was bounded by sand and had been heavily fished by industry producing a large amount of high quality roe.

In April 2001, ~1.4 t of sea urchin was transplanted form Barrens to Fringe with the co-operation of commercial divers (i.e. 5 person days). The density of sea urchins, and the yield and colour of their roe was sampled within each site until April 2002. At each of seven times, the density of sea urchins at each site was estimated using 10, 5 × 1 m transects placed haphazardly within each site, and 3 samples of 7 sea urchins were also collected. For each sea urchin, the total weight of the animal and the weight of roe were measured. The yield of roe for each sea urchin was then calculated as a proportion of the total weight. The colour of roe was matched against a standard colour chart grouped into two categories (i.e. marketable or not). Density, yield and the proportion of individuals with roe of a marketable colour were compared at the end of the experiment in a t-test.

To estimate the commercial benefits associated with the transplanted sea urchins, two commercial fishers harvested all the transplanted sea urchins, and these were taken to a local processor. The catch was weighed and roe processed for commercial sale. This entailed processing roe through various washes and draining the end product to get an estimate of recovery relative to the landed weight. The processor also graded the catch of transplanted sea urchins, and other un-transplanted individuals from Fringe, and gave an indication of their relative commercial value.

3.3.3. Results

3.3.3.1. Yield of roe

There was no increase to the yield of roe in transplanted sea urchins (Figure 3.3.1a, Table 3.3.1) after 6 weeks of the small-scale experiment. Yield from transplanted sea urchins was on average 3.3%, and was not significantly greater than that in the controls. This was much less than yield from sea urchins occurring naturally in Fringe, which were on average 13.7%.

In the medium-scale experiment, yield of roe was significantly higher in transplanted sea urchins after three months (Figure 3.3.2a, Table 3.3.2). Yield from sea urchins transplanted at low and

medium densities was significantly higher than yield at the highest density, which was significantly higher than sea urchins in Barrens (SNK, P <0.05). Yield from sea urchins transplanted at low and medium densities was not significantly different to yields from un-transplanted sea urchins occurring naturally in Fringe (SNK, P >0.05). In addition, there was no significant difference in yield of roe among control treatments (Figure 3.3.2a, SNK, P >0.05).

In the large-scale experiment, yield of roe was significantly higher in transplanted sea urchins after 12 months (Figure 3.3.3a; t-test, P <0.05). Immediately following the transplant in April 2001, the yield of transplanted sea urchins was not significantly different to yield from sea urchins in the three control sites (SNK, P >0.05). By May 2001, yield in transplanted sea urchins was higher than in control sites. Yield from sea urchins in all treatments declined over the spawning period (June-October 2001), and increased in the following recovery phase (see also Byrne *et al.* 1998). By April 2002, yield from the transplanted sea urchins was significantly higher than control sites (t-test, P <0.05, Figure 3.3.4a).

3.3.3.2. Colour of roe

Although differences in the proportion of individuals with roe of a marketable colour were similar to those for yield, there were no significant differences among treatments in the small- and medium-scale experiments. In the small-scale experiment, the proportion of sea urchins with roe of a marketable colour was not significantly different among treatments, and was half that observed for un-transplanted sea urchins in the Fringe (Figure 3.3.1b, Table 3.3.1). In the medium-scale experiment, the treatment with the highest proportion of individuals with roe of a marketable colour was the lowest density (Figure 3.3.2b). This was higher than in the Fringe and Barrens.

In the large-scale experiment, the proportion of sea urchins with roe of a marketable colour was higher in the transplant site than the controls (Figure 3.3.3b; t-test, P <0.05). At the start of the experiment, there was no significant difference between the transplanted individuals and controls (SNK, P >0.05). That is, the proportion of sea urchins with roe of a marketable colour declined significantly during the experiment. This occurred at two of the three control sites, where the proportion of individuals at the end of the experiment had declined to 14% and 33%. At the third control site, the proportion remained high at 86% and not significantly lower than the transplant site with 100% (SNK, P >0.05).

Table 3.3.1. Summary of analysis of variance in yield and proportion of individuals with roe of a marketable colour for sea urchins in treatments in the small-scale experiment. An * shows significant effects (P <0.05), and [a] = arcsine transformation.

Density	**Yield**			**Colour** [a]		
Source	**df**	**MS**	**F**	**df**	**MS**	**F**
Treatment	3	0.92	0.79	6	0.25	0.59
Res	80	1.17		14	0.42	
Total	83			20		

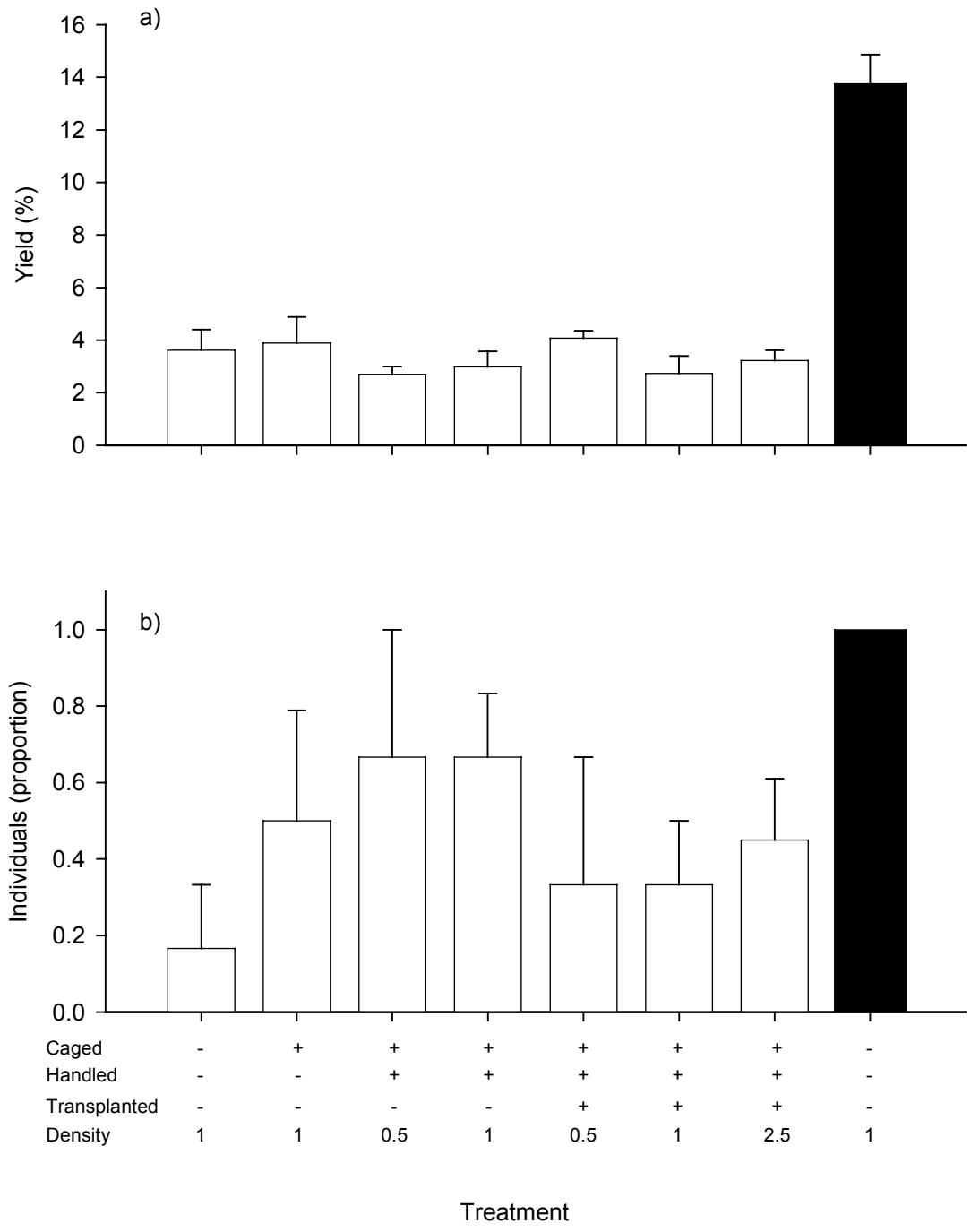

Figure 3.3.1. a) Yield of roe (+ SE) and b) proportion (+ SE) of individuals with roe of a marketable colour within different treatments in the small-scale experiment. Values for individuals in adjacent Fringe is also shown as a closed bar.

Table 3.3.2. Summary of analysis of variance in yield, proportion of individuals with roe of a marketable colour, and mortality, for sea urchins in treatments in the medium-scale experiment. An * shows significant effects (P <0.05), and [a] = arcsine transformation.

Roe	Yield			Colour [a]		
Source	**df**	**MS**	**F**	**df**	**MS**	**F**
Treatment, Tr	4	169.66	11.49 *	4	0.29	2.41
Res	10	14.77	3.40 *	10	0.12	1.67
Total	300			30		

	Mortality		
Source	**df**	**MS**	**F**
Treatment, Tr	2	121.04	6.02 *
Res	6	20.10	
Total	8		

Table 3.3.3. Summary of analysis of variance in yield, and proportion of individuals with roe of a marketable colour for sea urchins in treatments investigating artifacts of transplanting in the medium-scale experiment. An * shows significant effects (P <0.05), and [a] = arcsine transformation.

Roe	Yield		
Source	**df**	**MS**	**F**
Treatment, Tr	2	0.24	0.24
Cage (Tr)	6	1	1.22
Res	9	0.82	
Total	17		

	Colour [a]		
Source	**df**	**MS**	**F**
Treatment, Tr	2	0.07	0.33
Res	6	0.21	
Total	8		

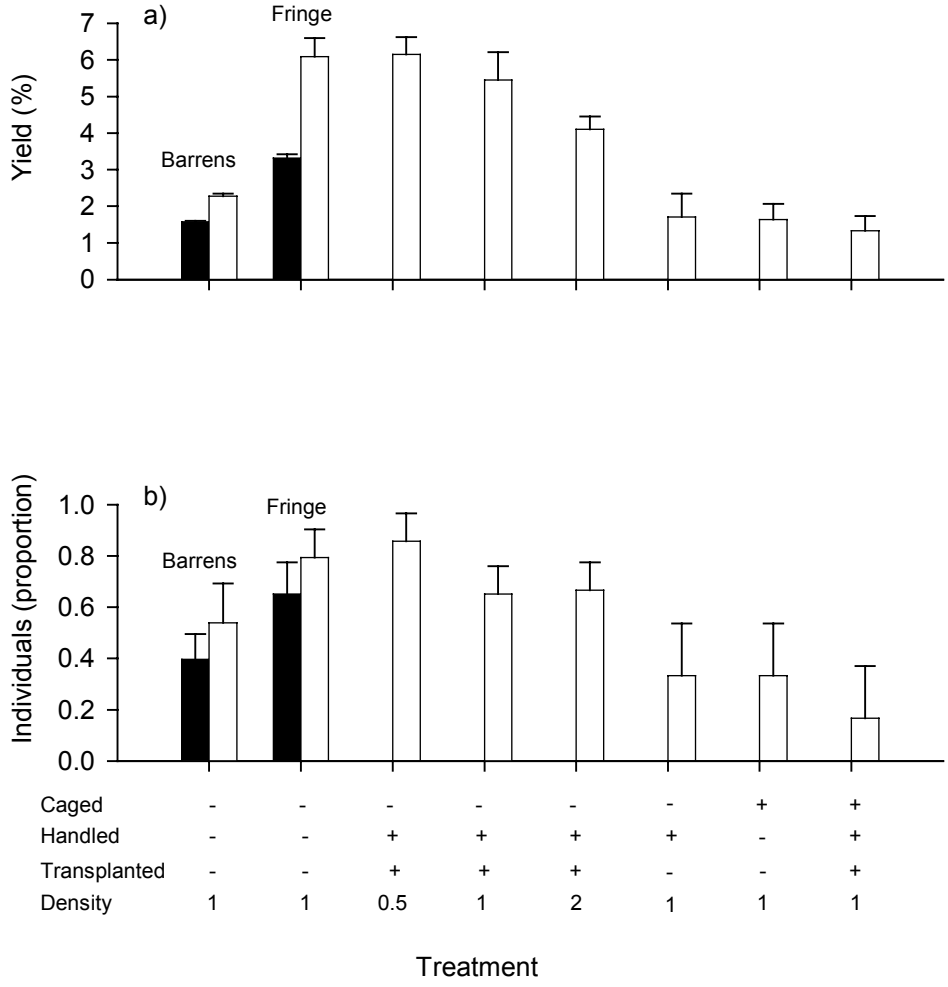

Figure 3.3.2. a) Yield of roe (+ SE) and b) proportion (+ SE) of individuals with roe of a marketable colour within different treatments in the medium-scale experiment. Closed bars represent samples taken before transplants and open bars after transplants. Values for individuals in adjacent Fringe and Barrens are also shown.

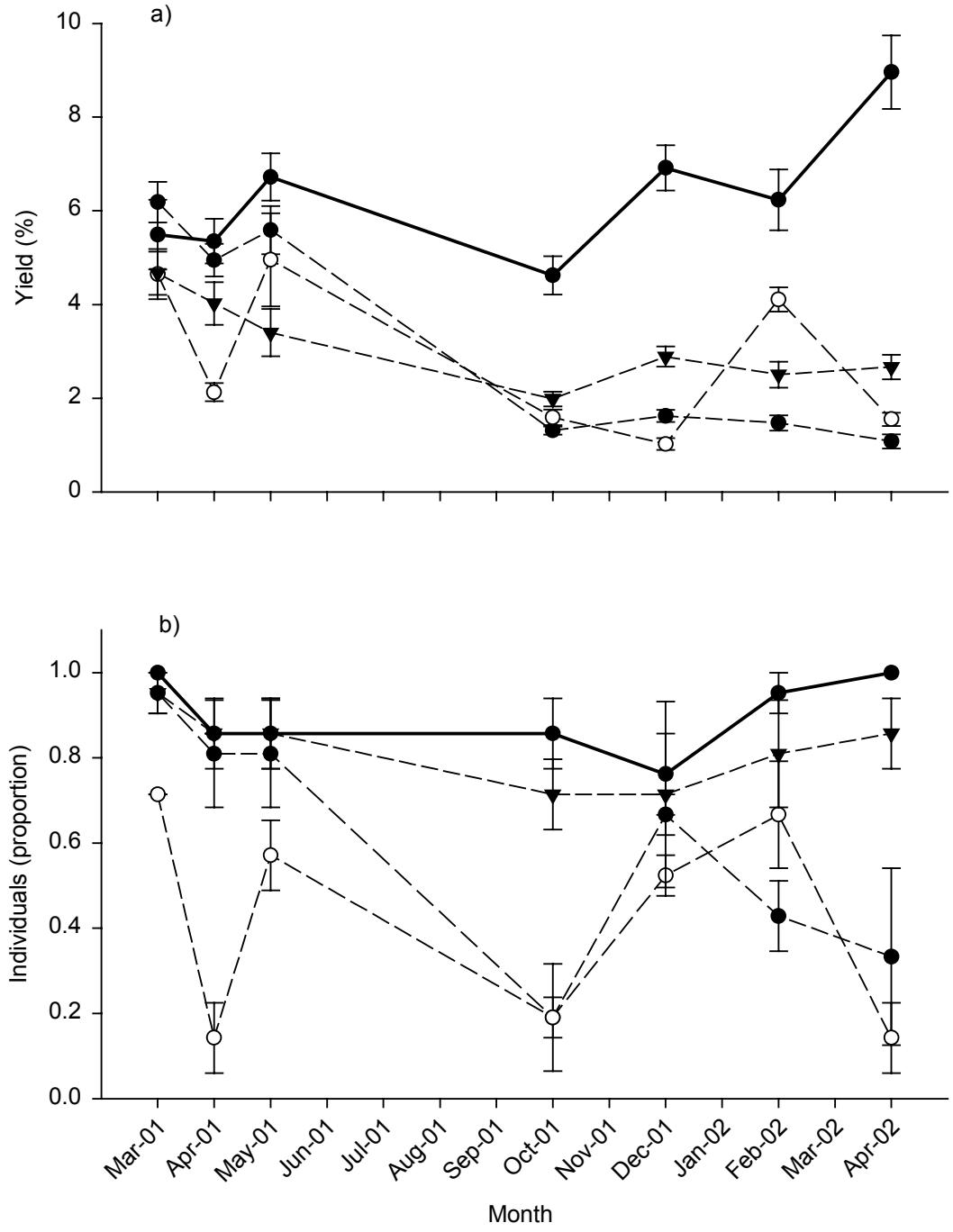

Figure 3.3.3. a) Yield of roe (± SE) and b) proportion (± SE) of individuals with roe of a marketable colour before and after different treatments in the large-scale experiment. The solid line represents the site individuals were transplanted to, whilst the dashed lined represents controls. Note, the first sample at each site was before the transplants.

3.3.3.3. Loss or mortality of individuals

In the medium-scale experiment, there were significantly different numbers of sea urchins lost from the different density treatments (Figure 3.3.4, Table 3.3.2), and these losses are most likely to be attributed to mortality. A significantly greater proportion of sea urchins were lost from the high density treatments than low and medium density (SNK test, P <0.05). In the large-scale experiment, the density of transplanted sea urchins declined rapidly from 15.0 m^{-2} in April 2001 to 7.6 m^{-2} by May 2001 (Figure 3.3.5). By October 2001, the density of transplanted sea urchins had declined to 4.0 m^{-2}, but remained at about 3.3 m^{-2} until the end of the experiment. Losses may have been caused by factors other than mortality. Although industry was asked not to fish in the transplant site, anecdotal evidence suggests some catch was removed. Further, during sampling of the highly aggregated population immediately following the transplant, there appeared to be several biases that may have increased the apparent density of individuals. Density in the controls varied among sites, but remained relatively consistent throughout the experiment (Figure 3.3.5).

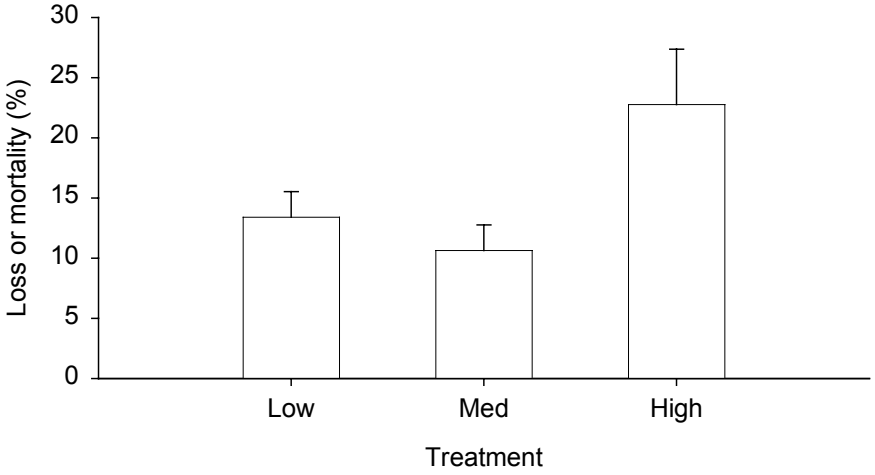

Figure 3.3.4. Loss or mortality (+ SE) of individuals from different treatments after transplanting in the medium-scale experiment.

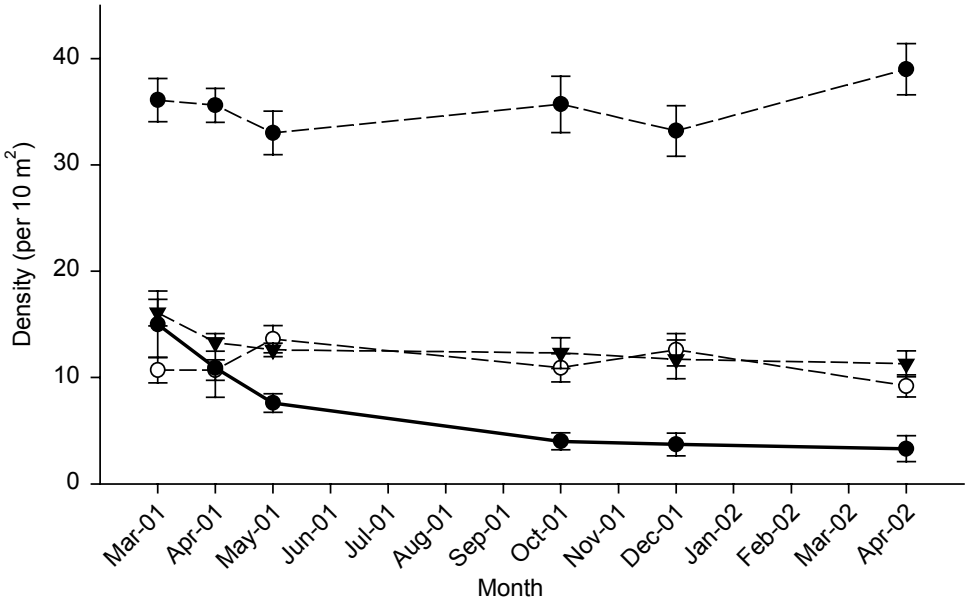

Figure 3.3.5. Density (± SE) of individuals before and after different treatments in the large-scale experiment. The solid line represents the site individuals were transplanted to, whilst the dashed lined represents controls.

3.3.3.4. Costs and benefits of transplanting

At the end of the large-scale experiment, commercial fishers harvested all of the sea urchins remaining in the transplant site. The total weight of sea urchins was 815 kg, or 58% of the initial weight of sea urchin that was transplanted. Before processing, the average yield of roe from transplanted sea urchins was 9.0% $\pm$ 0.8% of the total weight of individuals, but after processing this dropped to 6.2%. These yields were similar to those obtained from commercial catches of sea urchin at that time of year, but less than the maximum yields from experienced fishers (i.e. 10%). Over 40% of the roe from the transplanted sea urchins was considered by the processor to be of high quality. This is also similar to that obtained from commercial catches of sea urchin in Fringe at that time of year. A total of 50.5 kg of roe was recovered from the transplant site after processing. Combined with the current average market price of \$42 kg^{-1} this equates to a value of \$2100. The cost of processing and marketing the roe from sea urchins taken from the transplant site was estimated by the processor to be approximately \$1600, which included the cost of buying sea urchins from the commercial fishers (\$600), processing (\$600), and costs associated with transport and miscellaneous (\$200). As a result, the net profit to the processor was \$500.

3.3.4. Discussion

Centrostephanus are not usually caught by commercial fishers in Barrens because of the poor quality of their roe (Blount and Worthington 2002). We have shown that transplanting individuals to the Fringe can rapidly enhance the yield and colour of their roe. Similar enhancements in the yield and colour of sea urchin roe have been associated with an increase in the availability of food in aquaria and the wild (Russell 1998, Guillou and Lumingas 1999, Guillou *et al.* 2000). However, rapid enhancement has only been found for sea urchins in aquaria. Although reductions to the density of wild individuals within Barrens may also be used to enhance the roe of *Centrostephanus* (see Chapter 3.2), this technique is unlikely to be as rapid as transplants. Enhancements in roe following reductions in density are often delayed until after the benthic algal assemblage develops (see Chapter 3.2). An immediate greater availability of food within Fringe than Barrens, was probably the cause of the enhancements we observed after transplanting.

The commercial success of enhancing the roe of *Centrostephanus* by transplanting individuals from Barrens to Fringe depends on many factors. For example, the timing of the transplant must be considered, particularly if the objective is to enhance roe within a short period of time. Only limited enhancement was observed over short time periods or when the roe was not recovering after spawning. To maximise the speed of enhancement, transplants should be completed early in the recovery phase when roe is known to be accumulating nutritive material, and adding volume. For *Centrostephanus* this is between October and January (Byrne *et al.* 1998). We had much more success when individuals were transplanted in December, and significant improvements to yield and colour of roe of individuals were made in the short-term. In this case, yield and colour of roe was improved to levels similar to un-transplanted sea urchins living in Fringe.

As well as the timing of transplants, the density at which sea urchins are transplanted should also be considered. Significant improvements to yield were only observed when sea urchins were transplanted at densities equal to or less than the density at which sea urchins occur naturally in Fringe. Although yield was improved when sea urchins were transplanted at these densities, it was less than the yield from sea urchins occurring naturally in Fringe. Yield of roe of other species of sea urchin is also affected by density (McClanahan and Kurtis 1991). Density also effected the level of change in the roe from individuals within Barrens (See Chapter 3.2).

The density at which sea urchins are transplanted into Fringe may also effects their subsequent mortality. Our experiments showed a greater loss of transplanted sea urchins from higher densities.

Losses of 10-13% were observed when sea urchins were transplanted at densities equal to or less than that previously occupying the site. Losses of 23% occurred when sea urchins were transplanted at twice their original density. Given the limited opportunity for individuals to emigrate from the experimental sites, this suggests there may be increased mortality after transplants at higher densities. It is possible that shelter from predators and wave exposure is limiting at higher densities in Fringe (Andrew and Byrne 2001).

Transplanting sea urchins from Barrens to Fringe can significantly increase their market value over a short time period. Commercial fishers do not normally catch sea urchins in Barrens, and if caught, few can be marketed. Instead, fishers target the much smaller populations in Fringe that can be rapidly depleted. If it were cost effective, fishers could transplant some of the enormous populations in Barrens into Fringe. During our experiments, commercial processors were able to profit from processing and marketing the roe from transplanted sea urchins, and there was not much investment by fishers in time and money in transplanting. Further, the cost-effectiveness of transplanting will increase as Fringe populations become more depleted. Finally, the potential consequences of transplanting and manipulating the density of sea urchins also needs to be considered. For example, the enhancements to the marketability of roe (i.e. yield and colour) may also be related to a greater reproductive output. Also, because of their effect on algal populations, changes in the density of sea urchins can also effect many other invertebrate populations.

4. BENEFITS AND ADOPTION

This study has developed methods to assess the size of populations of sea urchins, and the quality of their roe, in NSW, eastern Victoria and Port Phillip Bay. A manual describing the sites used in the survey for each fishery is available from NSW Fisheries, and should enable them to be repeated in the future. The surveys provided estimates of the biomass of each population, which are likely to be close to unexploited levels because of limited catches to date. These estimates of biomass were combined with estimates of productivity to suggest potentially sustainable catches. Further, other recommendations about management of the fisherys were also made. This information has already been used in advice for determining appropriate TAC for one species in NSW, and is also likely to be used to determine TAC for each of the other fisheries surveyed. Techniques to improve the quality of roe for market have been developed and adopted by industry. The experiments involved have also provided some information about the effects of reduced densities of sea urchins on other species, and about the potential productivity of the sea urchin population. Further information about these issues will become available if the surveys are repeated in the future. All this information, together with an overview of the Japanese market for sea urchin roe, will continue to be used by managers and stakeholders as each fishery develops.

5. FURTHER DEVELOPMENT

This study has provided information about the size of populations of sea urchins, and the quality of their roe, in NSW, eastern Victoria and Port Phillip Bay. This information has already been used to recommend an appropriate TAC for one species in NSW, and is likely to also be used for all other species surveyed. Estimates of potentially sustainable catch are heavily dependent on the productivity of the population. Estimates of productivity have been made from experiments and by assuming rates of recruitment, growth and mortality. These assumptions can only be confirmed through future monitoring of the populations and fisherys. The surveys developed here should form a fundamental component of any future monitoring program. The frequency of any future survey should be related to development of the fishery. For example, those fisheries that are currently fully-developed (e.g. *H. tuberculata* in NSW) will require more frequent monitoring than other less exploited fisheries. The TAC Committee in NSW suggested surveys of *H. tuberculata* should be repeated after 5 years. Considering the value of the fishery (i.e. 60 t at $5 per kg) relative to the likely cost of survey (i.e. about $100 000), this seems an appropriate time-scale. On-going monitoring of the fishery may suggest surveys are required more frequently to address specific concerns (e.g. illegal catches).

Through the development of techniques to improve the quality of roe for market, this study has provided information about ways to increase the efficiency of the fishery through improvements in the productivity of the sea urchin population (i.e. greater return with reduced impact). The study also provides information about the likely effects of fishing on the sea urchin population and the associated assemblages of benthic algae and macro-invertebrates. The results suggested that the fishery can have localised effects on algal and macro-invertebrate assemblages and there is a need to consider management responses to minimise the ecological effects of fishing. More work is needed to fully understand particularly the response of macro-invertebrate populations to sea urchin fishing and how any effects of fishing can be minimised. The experiments that provided the information about techniques to improve roe quality and the effects of fishing were only completed for *Centrostephanus*, and should bo extended to the *Heliocidaris* spp. It is likely that use of similar techniques in the other fisheries may well increase efficiency and productivity, as well as providing information about the effects of fishing these species.

Finally, variable roe quality continues to complicate the further development of sea urchin fisheries in Australia. Field- and aquaria-based techniques can contribute to improvements in the quality of roe produced, but techniques of processing roe desperately need to be further developed. Some experiments already completed in NSW suggest enormous improvements could be made for a limited investment.

6. PLANNED OUTCOMES

The planned outcome of developing a process of stock assessment of sea urchins that can be integrated into an apropriate management framework for sea urchin fisheries in NSW and eastern Victoria has been achieved for red sea urchins in NSW and will soon be achieved in the other fisheries. This planned outcome has been achieved through two main outputs from the project. Firstly, reliable information about the size and productivity of populations has been used to set an appropriate TAC for red sea urchins in NSW, and similar information will be used to set TACs for other species being harvested. In addition, the production of a manual describing the sites used in the surveys for each fishery will allow similar surveys to be repeated at some time in the future. Another planned outcome was to provide a means of overcoming problems associated with a large proportion of populations of sea urchins having roe that is not of a high quality, which has been restricting development of sea urchin fisheries by reducing profit to divers and increasing processing costs. The project's outputs have contributed to this planned outcome by providing details on how to make harvesting of sea urchins more efficient using techniques that enhance the roe of individuals, and providing estimates of the cost-effectiveness of implementing these techniques. The final planned outcome was to develop an understanding of the impact of harvesting sea urchins, and techniques for enhancing roe, on habitat, and other species living with sea urchins, or associated with them. Along with a report by the Centre for Ecological Impacts of Coastal Cities the project reported on various aspects of this topic. However, more work will be required to comprehensively understand any impacts.

7. CONCLUSION

The project has been timely in that it has coincided with increased interest in exploiting sea urchins in NSW and Victoria. Consequently, outputs from the project have provided information that has been both relevant to the current status of these fisheries and valuable to their orderly development in the future. In addition to meeting objectives 1) developing and completing a process of stock assessment of sea urchins in NSW and eastern Victoria, and 2) investigating techniques to enable the reliable harvesting of quality roe from coastal reefs and determine their impacts on associated species, the project has provided an assessment of stocks of sea urchins in Port Phillip Bay and a market assessment of Australian sea urchin roe.

In order to meet objective 1, surveys were developed to estimate the density, size-structure, quality of roe, and biomass of sea urchins in NSW, eastern Victoria and Port Phillip Bay. With the limited development of these fisheries to date, and in combination with the market assessment of Australian sea urchin roe, this information was used to estimate the relative value of these fisheries in terms of the general quality of roe and the magnitude of sustainable catches. Outputs from this project have already been used by the Total Allowable Catch and Review Committee to set an annual TACC for red urchins in NSW, and it is anticipated that this will also be done for species in eastern Victoria and Port Phillip Bay. In addition, the production of a manual describing the location of sites used in assessments of sea urchins presented in this project allows for repeatability

of surveys and assessment of any changes to populations of sea urchins in response to future catches. This has already been done once successfully for red sea urchins in NSW.

Experiments involving reductions in density and transplanting of individuals were used to meet objective 2. Both techniques resulted in significant improvements to yield and colour of gonads of purple sea urchins in a relatively short period of time, and were determined to be a cost-effective means of making harvesting more efficient and enhancing the fishery for purple sea urchins. Furthermore, reductions in density may have some compensatory benefits to the population of purple sea urchins in terms of increased growth and recruitment. Reductions in density and transplanting could also be applied to other fisheries, in particular white sea urchins in Port Phillip Bay, where dense aggregations of individuals exist with low quality roe. However, these techniques were shown to cause changes to the assemblage of benthic algae, and implementation needs to be managed appropriately. In addition, surveys of macro-invertebrates living beneath purple sea urchins across most of their distribution in NSW, found over 100 taxa. More work is needed to determine the extent of the displacement of these species when sea urchins are harvested, in terms of their flexibility in habitat requirements.

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APPENDIX 1: Intellectual property

No patentable inventions or processes have been developed as part of this project. The work reported in this report will be published in scientific journals.

APPENDIX 2: Staff

Staff directly employed on this project with FRDC funds were:

Craig Blount Scientific Officer (NSW Fisheries)

Staff who contributed to the project but were not directly funded by FRDC were:

Dr Duncan Worthington	Principal Investigator (NSW Fisheries)
Dr Harry Gorfine	Co-Investigator (MAFRI)
Kane Organ	Fisheries technician (NSW Fisheries)
Peter Gibson	Fisheries technician (NSW Fisheries)
Ben Stewart	Fisheries technician (NSW Fisheries)
Rowan Chick	Fisheries technician (NSW Fisheries)
Cameron Dixon	Biologist(MAFRI)

Appendix 3: Subzones of the NSW Abalone and SUTS fishery

Catch Subzones for the NSW SUTS Fishery

A Tweed Heads to Ballina

B1 Ballina to Sandon
B2 Sandon to Red Rock
B3 Red Rock to Coffs Harbour

C1 Coffs Harbour to SW Rocks
C2 SW Rocks to Pt Macquarie

D1 Pt Macquarie to Harrington
D2 Harrington to Tuncurry

E2 Foster to Seal Rocks
E3 Seal Rocks to Hawks Nest Beach
E4 Yacaaba Head and Islands
E1 Broughton Island

F1 Port Stephens to Anna Bay
F2 Anna Bay to Newcastle
F3 Newcastle to Burwood Beach
F4 Burwood Beach to Swansea

G1 Swansea to Norah Head
G2 Norah Head to The Entrance
G3 The Entrance to Terrigal
G4 Terrigal to Broken Bay

H1 Broken Bay to Sydney Harbour
H2 Sydney Harbour to Bondi Beach
H3 Bondi Beach to Botany Bay

J1 Botany Bay to Port Hacking
J2 Port Hacking to Marley Beach
J3 Marley Beach to Garie Beach
J4 Garie Beach to Stanwell Park
J5 Stanwell Park to Wollongong Hbr

K1 Wollongong Hbr to Shellharbour
K2 Shellharbour to Bombo Beach
K3 Bombo Beach to Werri Beach
K4 Werri Beach to Shoalhaven Heads

L1 Shoalhaven Heads to Currarong
L2 Currarong to Pt Perpendicular
L3 Inside Jervis Bay
L4 Nth tip Bowen Island to Wreck Bay

M1 Wreck Bay to Bendalong
M2 Bendalong to Ulladulla

N1 Ulladulla to Termeil Point
N2 Termeil Point to Brush (excl. Island)
N3 Brush Island

P1 Brush (excl. Island) to Pretty Beach
P2 Pretty Beach to Sth Durras
P3 Sth Durras to North Head
P4 North Head to Batemans Bay

Q1 Batemans Bay to Lilli Pilli Beach
Q2 Tollgate Islands
Q3 Lilli Pilli Beach to Malua Bay
Q4 Malua Bay to Burrewarra Point
Q5 Burrewarra Point to Moruya River

R1 Moruya River to Black Rock
R2 Black Rock to Tuross Lake

S2 Tuross Lake to Dalmeny
S3 Dalmeny to Narooma

S1 Montague Island

T1 Narooma to Corunna Lake
T2 Corunna Lake to Bermagui

U1 Bermagui to Cuttagee Inlet
U2 Cuttagee to Thibbul Inlet (Murrah)
U3 Thibbul to Bunga Beach (Goalen and Pressure)
U4 Bunga Beach to Mimosa Rocks (Bunga)

V1 Mimosa Rocks to Bithry Inlet
V2 Bithry Inlet to Barounda Inlet
V3 Barounda Inlet to Tathra

W1 Tathra to Wallagoot Lake
W2 Wallagoot Lake to Short Point Beach
W3 Short Point Beach to Merimbula

X1 Merimbula to Long Beach
X2 Long Beach to Eden Wharf

Y11 Eden Wharf to Red Point
Y12 Red Point to Leatherjacket Beach
Y13 Leatherjacket Beach to Mowarry Point

Y21 Mowarry Point to Saltwater Beach
Y22 Saltwater Beach to Long Point
Y23 Long Point to Bittangabee Bay
Y24 Bittangabee Bay to Green Cape

Y31 Green Cape to City Rock
Y32 City Rock to Wonboyn

Z1 Wonboyn to Jane Spiers Beach
Z2 Jane Spiers Beach to Black Head Anchorage
Z3 Black Head Anchorage to Nadgee Lake
Z4 Nadgee Lake to Howe Beach
Z5 Cape Howe

APPENDIX 4:

Effects of manipulating density of *Centrostephanus rodgersii* on benthic algae in Barrens and Fringe in New South Wales, Australia

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Introduction

Herbivores have a large influence on the abundance and species composition of plant communities in both terrestrial and marine systems. In general, grazing in marine systems is approximately three times greater than in terrestrial systems, and in some instances herbivores may remove up to 100% of primary production (Cyr and Pace 1993). In temperate marine systems grazing by sea urchins has a significant impact on the abundance of algae and species composition within algal assemblages, and in many cases, sea urchins are able to create and maintain areas devoid of foliose algae termed Barrens (Fletcher 1987, Himmelman 1983, Paine 1969, Andrew 1989, Andrew 1993, Andrew 2000).

Most of what is known about the impact of sea urchins on algal assemblages has come from studies where sea urchins have been completely removed from Barrens. After removals, there is generally a progression from crustose coralline algae to filamentous and finally foliose species, and the assemblage generally becomes dominated by a single canopy forming species such as *Sargassum* or *Ecklonia radiata* in the Southern Hemisphere (Andrew 1993) or *Alaria* in the Northern Hemisphere (Himmelman 1983). Complete removals of sea urchins do not consider the effect that density may have on the grazing patterns of sea urchins, and consequently the effects of sea urchin density on the abundance and composition of algal assemblages. Experiments involving partial removals can address these issues but to date have been rare in North America and only recently considered in the Mediterranean and Australasia (Andrew 1993, Benedetti-Cecchi 1998). Furthermore, little is known about the processes involved in the creation of Barrens, as removal experiments are not designed explicitly to answer this question. An understanding of the processes involved in the transformation of an area dominated by macroalgae to Barrens requires sea urchin densities to be increased in these areas. Few studies have attempted this (e.g. Andrew 1993) and none have done so at a range of densities.

Recent seabed mapping in southeast Australia (Andrew 2000) has determined that nearshore reefs in NSW contain as much as 50% Barrens. The creation and maintenance of these Barrens is largely due to the grazing of a single species of sea urchin, *Centrostephanus rodgersii* (Fletcher 1987), which may be found from the central NSW coast to Tasmania (Jones 1990). During daylight hours, *Centrostephanus* occupies crevices. At night, individuals emerge to graze, sometimes travelling up to 5 m from the crevices to forage (Andrew 1989, Andrew 1993, Fletcher

1987), and in doing so form a halo of Barrens around crevices that abruptly changes into habitat dominated by turfing and foliose algal species, or Fringe (Underwood, 1991).

A small sea urchin fishery based on *Centrostephanus rodgersii* and *Heliocidaris tuberculata* is developing in NSW. Currently only about 50 tonnes of *Centrostephanus* is harvested per year (Blount 2002) and inconsistency in yield, and poor roe quality, are some of the problems restricting the development of this fishery. To address these issues, the manipulation of sea urchin densities, in both Barrens and Fringe habitats, is being investigated as a means for improving yield and quality of roe in wild sea urchins (Blount 2002). If these techniques are to become widely used it is important to understand the impact of density manipulations on algal assemblages.

The purpose of this study was to develop an understanding of the processes involved in the transition between areas dominated by macroalgae to Barrens, and vice-versa, and to investigate the role of density. To achieve this, three separate experiments were undertaken. Firstly, sea urchins were caged at several densities in Fringe habitat, and secondly, sea urchins were caged at several densities in Barrens. In addition, in a complementary experiment to the Fringe experiment, *Centrostephanus* were offered species of algae that were prominent in the Fringe to investigate whether feeding preference determined the rate of loss or proliferation of algal species in the Fringe.

Methods

Study Sites

Caging experiments in Fringe and Barrens were carried out at the entrance to Port Hacking, Sydney, Australia. The Fringe experiment was located at Salmon Haul Bay on a 10m wide rocky platform at a depth of approximately 2 m. The algal assemblage consisted of juvenile *Sargassum vestitum, Sargassum linearifolium, Ecklonia radiata, Zonaria diesingiana, Phylospora comosa,* encrusting coralline algae, *Corallina officianis, Petalonia fascia, Champia compressa*, and *Amphiroa anceps* and purple sponge (in decreasing abundance). The Barrens experiment was located at Jibbon Head at a depth of approximately 5 m, where the natural density of sea urchins was estimated to be approximately 4 m^{-2}. At this site crustose coralline algae was dominant and foliose algae absent. The feeding preference experiment was located at Long Bay, in Barrens at a depth of approximately 5 m.

Fringe experiment

Sea urchins were contained in 0.49 m^2 galvanised steel mesh cages (70 x 70 x 20 cm) that were bolted to the sea floor. Sea urchins densities were caged in four treatments: 0%, 50%, 100% and 250% of natural Barrens density, corresponding to 0, 1, 2, and 5 individuals per cage, respectively. Control plots of equal area were also marked out on the rock platform to test for caging artefacts. Each treatment consisted of three replicate plots or cages. Although sea urchins in Barrens are known to be able to forage up to 5m from crevices, it was expected that they would graze progressively outwards from the edge of the cages (which essentially form artificial crevices) towards the centre. Hence, sampling was stratified into centre and edge of cages and two replicates taken at each position.

Cages were sampled approximately every two weeks from April until August 2001. The percent cover of each species (see study site and organisms) was recorded using a 20cm square quadrat with a 2.5 cm grid. At the beginning of the experimental period total algal cover, percent cover of foliose algae, *S. vestitum,* and *S. linearifolium* did not differ among treatments, although uncaged control plots contained more crustose algae than other treatments. At the final sampling date photographs were taken of the entire plot and the percentage grazed recorded, and compared with a

one-factor analysis of variance to determine if there were any significant differences between treatments.

Total percent cover of algae, the cover of crustose algae, *S. vestitum, S. linearfolium* and *Z. diesingiana* at each sampling date were compared in a three factor, mixed model analysis of variance. Treatment was considered a random factor, Plot was considered a random factor nested within Treatment, and Position fixed. The assumptions of normality and homogeneity of variance were checked by examining histograms of residuals and scatterplots of residuals versus means (Quinn and Keough 2001). Rare species were not analysed because they could not be transformed to meet assumptions of analysis of variance. Significant treatment effects were further analysed by Tukey's post hoc comparison tests to determine which treatments differed.

The composition of the algal assemblage was contrasted among Treatments using multidimensional scaling and a two factor analysis of similarity. Data was square root transformed in order to increase the weighting of rare species (Clarke 1994) and the Bray–Curtis similarity matrix was used for both multidimensional scaling and analysis of similarity. Differences among treatments were tested for using a two factor analysis of similarity where Plots were nested within Treatments. It was not possible to include the third level of the experiment, Position, because of limitations of the multivariate statistics package Primer. Therefore, Treatment and Plot were analysed because in univariate analyses these 2 factors were often significantly different, whilst Position within the cages was rarely so.

Barrens experiment

To test the influence of *Centrostephanus* grazing on the algal assemblage in the Barrens, sea urchins were contained in 1.25 m^2 galvanised steel mesh cages, 1.5 x 1.5 x 0.2 m) that were bolted to the sea floor. Four Treatments were established: 0%, 33%, 66%, and 100% natural density, corresponding to 0, 3, 6, and 9 individuals per cage, respectively. Uncaged plots of equal area were marked out on the seafloor to test for the effects of caging and each treatment contained three replicate plots or cages. Sampling in the Barrens was also stratified into two positions (centre and edge) within each cage, and two replicates taken at each position.

Sampling involved taking photographs every 1 to 2 months with a Sea & Sea™ underwater camera fitted with a macro-lens, flash, and photo frame. Since most species in the Barrens were small, being recent recruits, or encrusting forms, photographs provided an accurate representation of the algal assemblage at the time, whilst reducing dive time. Percent cover of each algal species was measured by placing an overlay of random points on the photograph and then identifying each species. Total percent cover of crustose coralline algae, juvenile *Sargassum vestitum*, and filamentous red algae were compared using analysis of variance, where factors and post-hoc comparisons were the same as for the Fringe experiment. Multidimensional scaling and a two factor analysis of similarity were also used to test for differences in the algal assemblage between Treatments. This site was characterised by crustose coralline algae and bare rock with a scattering of limpets, barnacles and anemones and at the beginning of the experiment, and the percent cover of crustose coralline algae and filamentous algae did not significantly differ between treatments.

Feeding preference experiment

In a multi-algal feeding preference experiment at Long Bay, four species of algae were fed to *Centrostephanus*. These were *Ecklonia radiata, Sargassum vestitum, Sargassum linearifolium* and *Zonaria diesingiana*, as these were the dominant foliose species in Fringe.

Trials involved placing one sea urchin of approximately 80-90 mm diameter in each of ten 20 litre plastic buckets, covered with a plastic mesh lid. Approximately 5-6 g of each algal species was then fixed to a weight and placed on the bottom of each bucket. To account for autogenic changes

in the mass of the algae (Steinberg and van Altena 1992), another 5-6 g of algae was placed inside a smaller plastic basket that floated above the sea urchins.

Sea urchins were acclimatised to the buckets for one week prior to the trials and were fed *ad libitim* on *Sargassum linearifolium.* After this period the weighed samples of algae were fed to the sea urchins which were allowed to graze for 4-5 nights. A total of 3 trials were conducted, with trials 1 and 3 lasting four nights, while trial 2 lasted for five nights. Sea urchins were not starved at any time in the trials in order to simulate the availability of food in the Fringe.

As each trial showed the same trend in food preference irrespective of the length of the trial, results were pooled and a paired t-test was carried out between each algae species to determine if grazing had occurred (as opposed to autogenic changes). Since there was more than one species of algae available to the sea urchins in each bucket replicates were not independent, therefore a one factor analysis of variance using six replicates of each species randomly chosen from the pooled data (i.e., from different buckets in each trial) was used to analyse the mass lost between species.

Results

Fringe experiment

High densities of sea urchins in the Fringe had significant impacts on the abundance of algae and the species composition of the algal assemblage. The total cover of algae, which consisted of all foliose and encrusting species, increased slightly over the 19 weeks in which the experiment was run in all treatments except where sea urchins were caged at 250% natural density. In this treatment, after 7 weeks, grazing caused larger patches of bare rock to appear within plots, and total cover began to decline (Figure 1a). Significant differences in total cover among Treatments occurred at weeks 9, 14 and 19 (conclusion of the experiment). At week 9 and 14 sea urchins at 250% natural density had significantly less total cover than controls, and by the conclusion of the experiment total cover in this treatment was significantly less than for all other treatments (Table 1). At all sampling dates there was no significant effect due to position within the cage, although Plots were often significantly different from each other. At the conclusion of the experiment, photographs taken of entire plots indicated that the percent area grazed was significantly different among Treatments (F= 8.89, df= 2,5; P= 0.023) and that sea urchins at 250% natural density had a larger proportion of grazed area those at 50% and 100% natural density (Figure 2).

S. vestitum was the dominant alga in most plots, and grew rapidly during the course of the experiment as seen in control treatments where the percent cover of this species increased from 20% to approximately 80% by the end of the experiment (Figure 1b). In the 50% and 100% natural density treatments cover of *S. vestitum* increased until week 14, but steadily decreased where sea urchins were at 250% natural density. At the final sampling date uncaged plots, and caged plots without sea urchins, both contained significantly more *S. vestitum* than where sea urchins were kept at 250% natural density, whilst sea urchins kept at 50% and 100% natural density were not statistically different to any of the treatments (Table 1a).

Although many species initially recorded in plots in low abundances became even rarer over time, grazing did not have a large impact on the percent cover of *S. linearifolium, Zonaria diesingiana* or crustose algae in plots. The average cover of *S. linearifolium* (Figure 1c) and *Z. diesingiana* was generally less than 20% in all treatments and differences between treatments in the cover of *S. linearifolium* were never significant (for example Table 1a). The cover of crustose algae, which included encrusting coralline algae and *Ralfsia*, was more erratic (Figure 1d). Initially the cover of crustose algae in the control treatment without cages was significantly higher than in the other treatments (F=4.62, df=4, P=0.02), but by week 4 had declined to similar levels to the other

treatments. At all other sampling dates there were no significant differences between Treatments or Position within cages, although Plots were significantly different (Table 1).

The effect of sea urchin grazing on the composition of the algal assemblage in the Fringe took longer to manifest than the impact on individual species such as *S. vestitum*. Initially the composition was the same across all treatments (ANOSIM, R=0.147, P=0.085) despite the fact that control plots without cages had a greater cover of crustose algae. The changes in the average community composition of each treatment through time is represented by the MDS plot in Figure 3. As time progressed assemblages in cages without sea urchins, and those where sea urchins were kept at 50%, 100% and 250% natural density followed a similar composition vector, indicating that changes to assemblages followed a similar path. Only at the final sampling date were there significant differences between treatments in terms of the composition of the algal assemblage (R= 0.399, P=0.021), which appears to be largely due to sea urchins kept at 250% natural density.

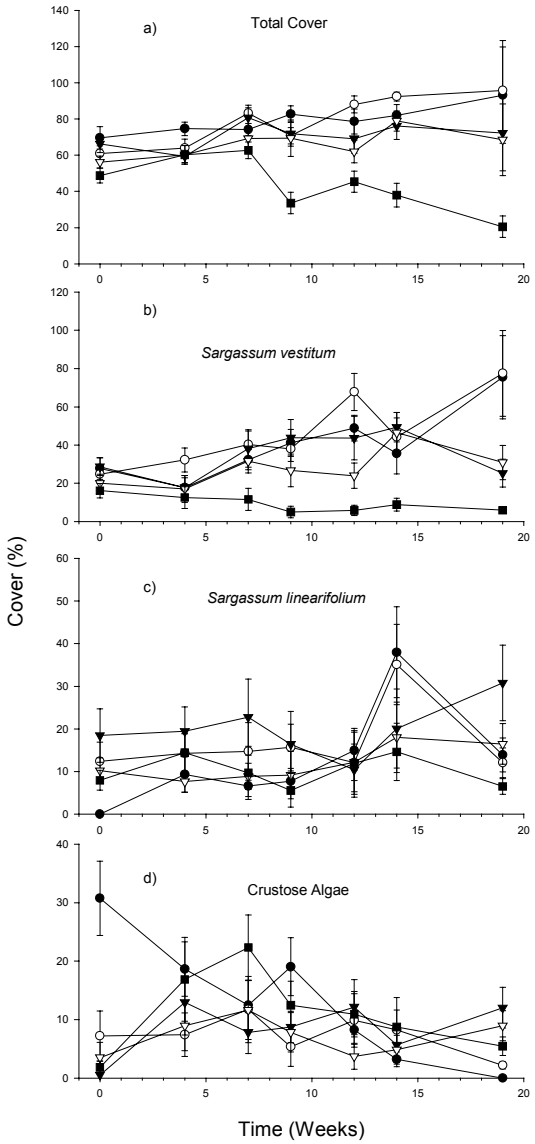

Figure 1. Percent cover (± SE) at each sampling time of a) total algae, b) *Sargassum vestitum*, c) *Sargassum linearfolium*, and d) crustose algae in 5 treatments in the Fringe experiment (●=uncaged control, ○=caged control, ▽=250% natural density,▼= 100% natural density, and ■= 250% natural density).

Table 1. Summary of analysis of variance in a) the total cover of algae, *Sargassum vestitum*, *Sargassum linearfolium* and crustose algae in the Fringe experiment and b) the cover of crustose algae, *Sargassum vestitum*, filamentous red algae and limpets in the Barrens experiment, at the last sampling period (week 19). An * shows significant effects ($P<0.05$), and ^{l} = log + 1 transformation.

a) Fringe		**Total cover**		***Sargassum vestitum***		***Sargassum linearfolium***		**Crustose algae**	
Source	**df**	**MS**	**F**	**MS**	**F**	**MS**	**F**	**MS**	**F**
Treatment, Tr	4	11133.91	17.49 *	12229.61	7.69 *	986.54	0.56	287.41	0.95
Position, Po	1	666.09	2.16	1676.91	2.81	0.41	0.00	68.69	0.75
Po*Tr	4	308.38	0.62	596.83	0.97	703.55	2.33	91.17	1.54
Plot, Pl(Tr)	10	636.65	2.41 *	1590.81	2.44 *	1774.17	6.31 *	302.04	5.98 *
Po*Pl (Tr)	10	497.01	1.88	612.84	0.94	302.29	1.08	59.39	1.18
Res	30	263.85		651.24		281.16		50.50	
Total	59								

b) Barrens		**Crustose algae**		***Sargassum vestitum*** ^{l}		**Filamentous red algae** ^{l}		**Limpets**	
Source	**df**	**MS**	**F**	**MS**	**F**	**MS**	**F**	**MS**	**F**
Treatment, Tr	4	5951.51	19.26 *	21.53	17.17 *	11.26	11.95 *	119.71	14.22 *
Position, Po	1	472.87	1.09	5.73	2.20	0.68	0.16	0.60	0.03
Po*Tr	4	435.74	6.52 *	2.60	2.35	4.20	2.86	17.64	1.21
Plot, Pl(Tr)	10	309.05	4.47 *	1.25	6.17 *	0.94	1.32	8.42	0.99
Po*Pl (Tr)	10	66.83	0.97	1.11	5.46 *	1.47	2.06	14.58	1.71
Res	60	69.10		0.20		0.71		8.53	
Total	89								

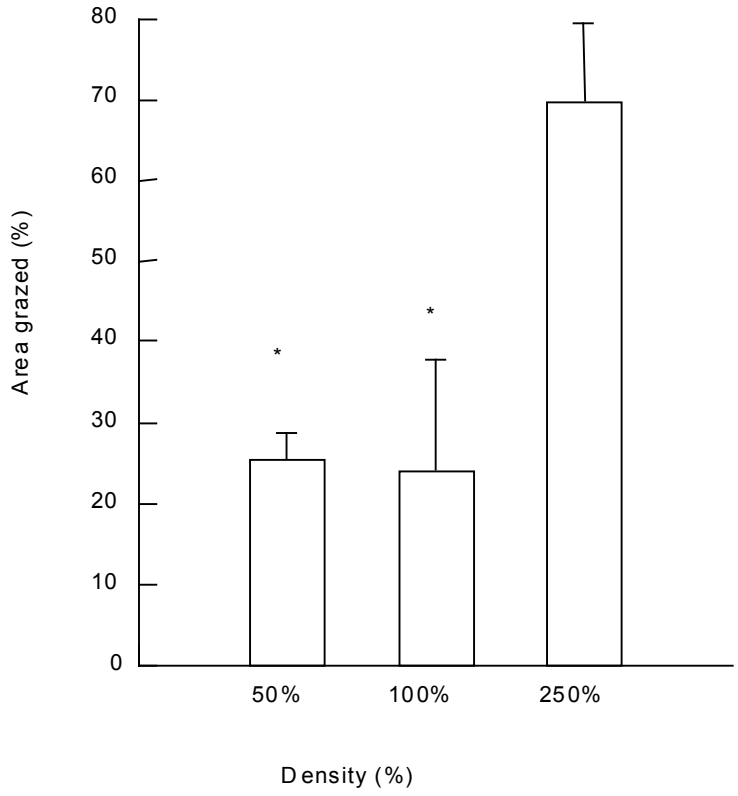

Figure 2. Area grazed (± SE) at the final sampling time (week 19) for treatments where the density of *Centrostephanus* was 50%, 100%, and 250% natural density of Barrens, in the Fringe experiment. An * indicates statistically similar in Tukey's post hoc analysis.

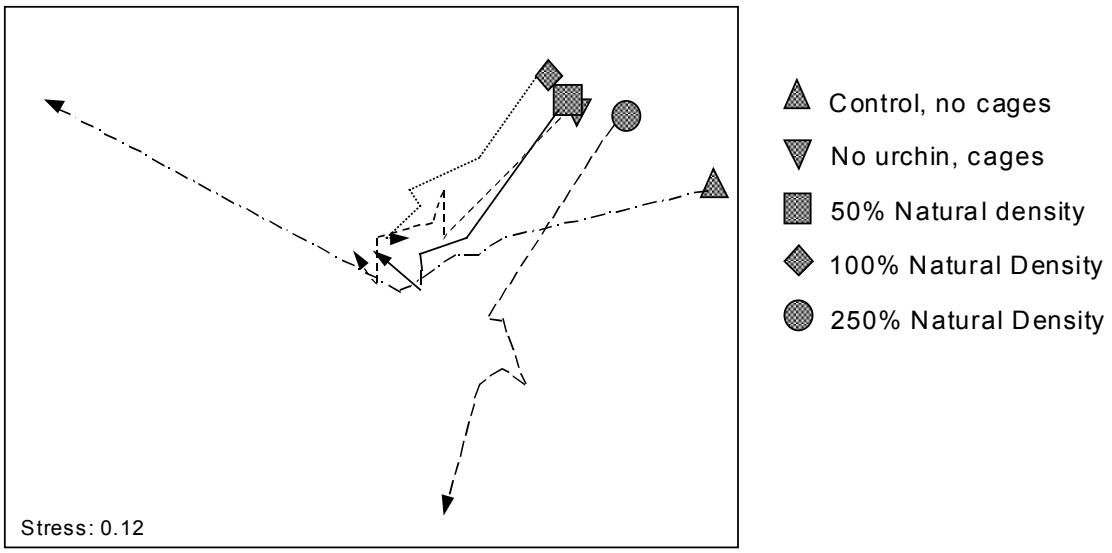

Figure 3. MDS plot representing the average assemblage for each treatment over the duration of the Fringe experiment. Symbols indicate the composition of the initial assemblage and arrows, the assemblage at the final sampling date.

Barrens experiment

All treatments began with between 65% and 80% crustose algal cover, but this decreased over time in all treatments except the control without cages (Figure 4a). After reductions were made the cover of crustose algae was proportional to sea urchin density at each sampling time. By week 8 the cover of crustose coralline algae in plots where there were no sea urchins and where sea urchins were kept at 33% natural density treatment was significantly different to the cover in plots where sea urchins were kept at 66% and 100% natural density (ANOVA, $P<0.05$, SNK). After 13 weeks the cover of crustose algae in plots containing no sea urchins was significantly less than the other three treatments (ANOVA, $P<0.05$, SNK). By week 22 the cover in plots where sea urchins were kept at 0%, 33% and 66% natural density was significantly less than the control treatment without cages (Table 1b, SNK). However, this result for week 22 needs to be interpreted with caution as there was an interaction between the effect of Treatment and Position within the cage due to plots where sea urchins were kept at 66% natural density containing less cover of crustose algae in the centre of plots.

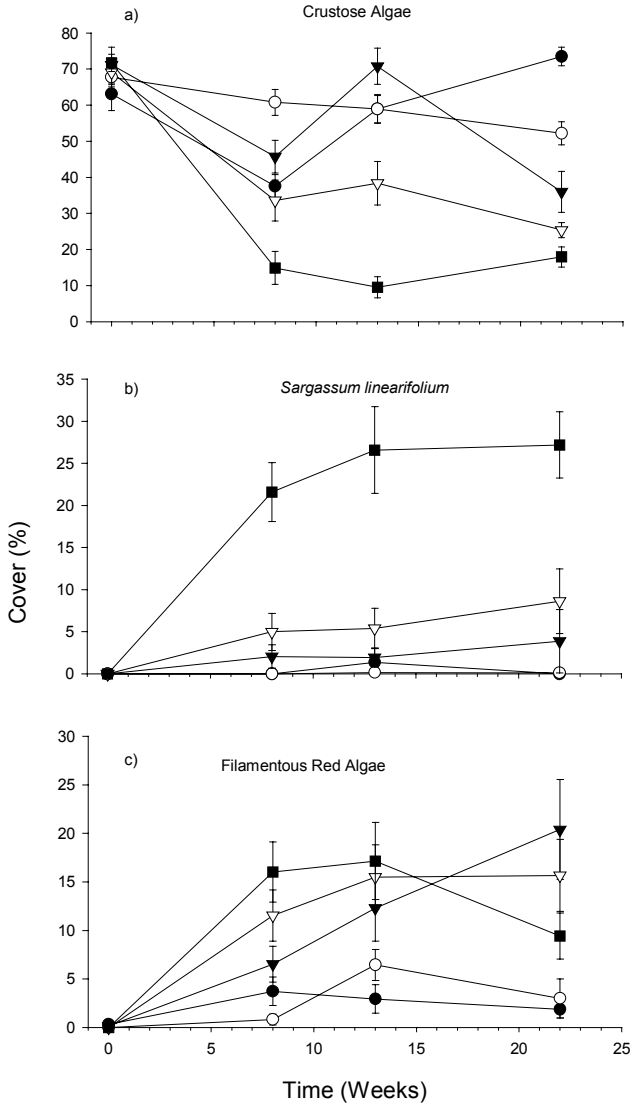

Figure 4. Percent cover (± SE) at each sampling time of a) crustose algae, b) *Sargassum vestitum*, and c) Filamentous red algae in 5 treatments in the Barrens experiment (●=uncaged, ○=100% natural density, ▼=66% natural density, ▽= 33% natural density, and ■= 0% natural density).

Reductions in the density of sea urchins resulted in an increase in the cover of *S. vestitum*. Within the first 8 weeks of the experiment (January and February) the pattern of recruitment of *S. vestitum* to reduced density plots was inverse to the density sea urchins (Figure 4b), and cover changed little after this time. Accordingly, from 8 weeks onwards the cover of *S. vestitum* in the complete removal treatment was significantly greater than all other treatments (Table 1b).

Plots where sea urchins were kept at natural density consistently supported a low cover of filamentous algae. Prior to week 13, cover of filamentous algae in the other treatments increased in proportion to sea urchin density so that only the natural density treatment with cages was not significantly different from cages without sea urchins (Table 1b, Figure 4c). After week 13 cover decreased in plots where there were no sea urchins, increased in plots where sea urchins were kept at 66% natural density and remained relatively constant in plots where sea urchins were kept at 33% natural density. At week 13 differences between treatments were not significant (ANOVA, $P<0.05$), although by week 22, sea urchins kept at 33% and 66% natural density contained significantly more filamentous red algae than natural density treatments (controls). Plots containing no sea urchins was not significantly different from the other treatments at the time (Table 1b).

During the first 8 weeks, assemblages of algae were similar across all treatments (Figure 5). As the experiment progressed the reduced density plots began to separate from control plots, the treatment without urchins showing the greatest degree of difference. By week 22 changes in the algal composition of the no urchin treatment were significantly different from the community composition of natural Barrens (ANOSIM $P<0.001$; $R= 0.699$).

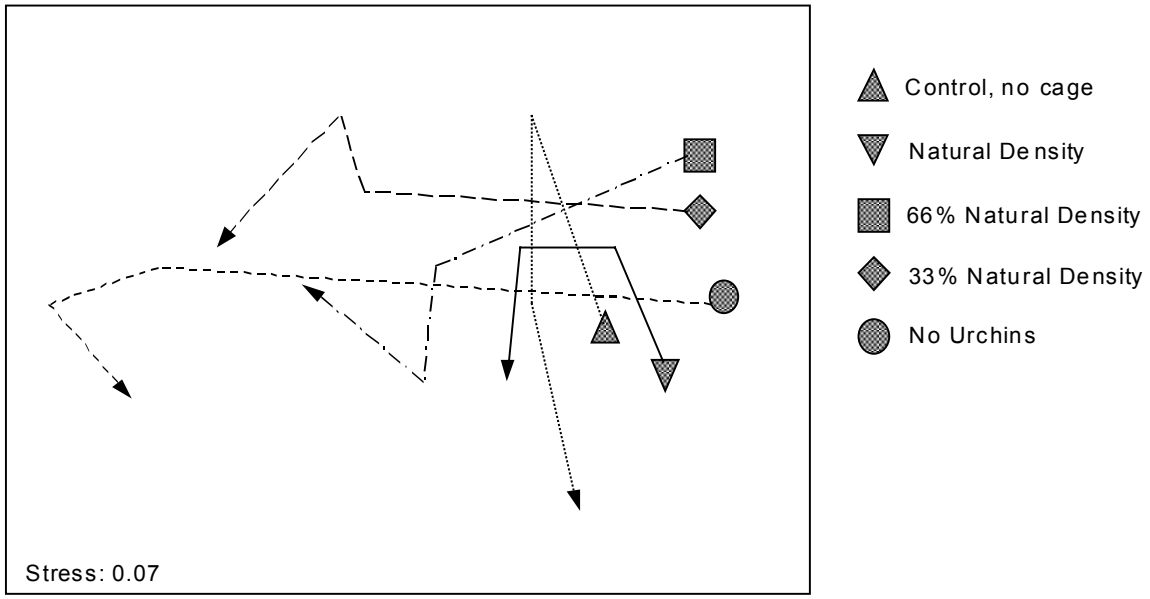

Figure 5. MDS plot representing the average assemblage for each treatment over the duration of the Barrens experiment. Symbols indicate the composition of the initial assemblage and arrows, the assemblage at the final sampling date.

Feeding preference experiment

Feeding preference assays indicate that *Centrostephanus* does shows preferences when presented with a range of algae. Paired t-tests confirmed that each species of algae was consumed as the mass loss in grazed pieces differed significantly from controls. *S. vestitum* was eaten significantly less than *E. radiata*, *S. linearifolium* and *Zonaria spp*. (F= 19.40, df= 3,20, P<0.001, Figure 6). On average each sea urchin consumed approximately 12.94g of algae in each trial, which equates to approximately 3.23g per day.

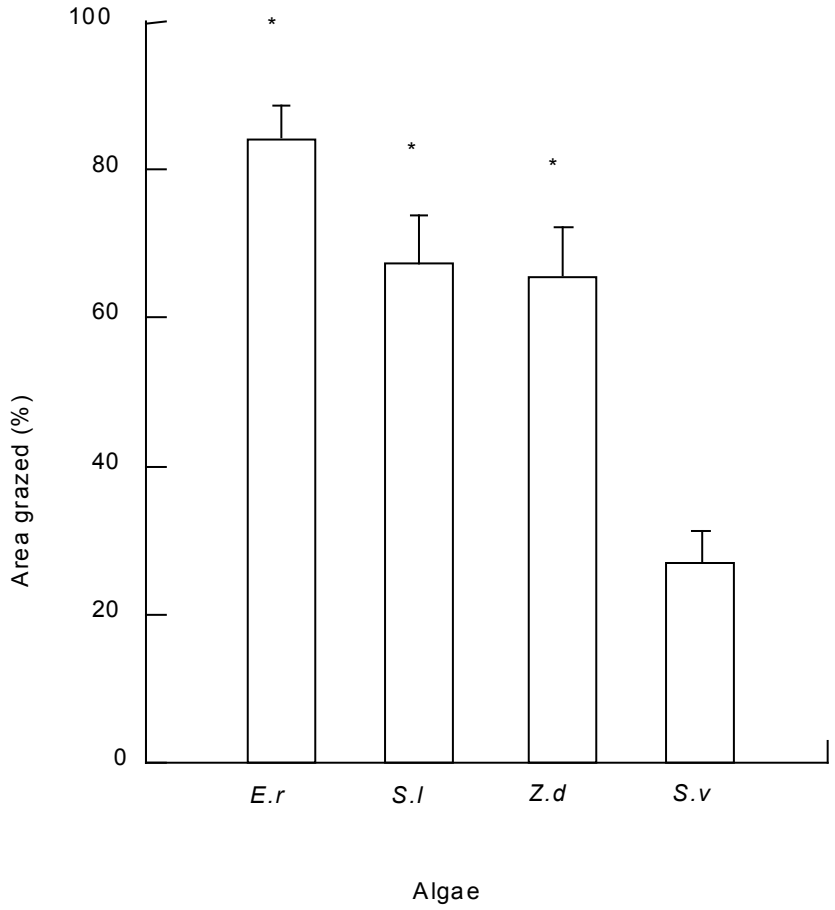

Figure 6. Alga consumed (± SE) in the feeding preference assay. *E.r = Ecklonia radiata, S.l = Sargassum linearfolium*, S.v = *Sargassum vestitum*, and *Z.d = Zonaria diesingiana* An * indicates statistically similar in Tukey's post hoc analysis.

Discussion

Reducing densities of *Centrostephanus* in Barrens encouraged the growth of filamentous red algae and, where *Centrostephanus* was completely removed encouraged recruitment of foliose algae to the detriment of crustose coralline algae. Only 33% of the natural density was required to maintain the Barrens free of foliose algae. In contrast, densities of *Centrostephanus* needed to be increased to 250% of natural density in Barrens to impact on assemblages of algae in Fringe.

Transition from Barrens to Fringe

It is well understood that the removal of sea urchins from Barrens results in a brief dominance of filamentous algae, followed by the proliferation of foliose algae and a decline in the cover of

coralline algae (Dean 1984, Himmelman 1983, Leinaas 1996, Paine 1969, Andrew 1991), and this also occurred in our study. Leinaas and Hartvig, 1996) found that experiments where sea urchins were almost totally removed encouraged the growth of kelps that outcompeted filamentous algae, and moderate reductions supported opportunistic species and no kelp. Similarly, in this study reductions to intermediate urchin densities led to an increase in the cover of filamentous algae and complete removals encouraged the recruitment of foliose algae.

Only 33% of the natural density was required to maintain Barrens free of foliose algae, a result that concurs with Andrew and Underwood, 1993). While the percent cover of some algal species at certain sampling dates seemed to correlate well with density, these relationships changed throughout the course of the experiment. In the case of *S. vestitum,* significant recruitment only occurred where there were no sea urchins. Therefore, our results support Andrew and Underwood's, 1993) conclusion that the impacts of sea urchin grazing are not linearly related to density.

In the majority of studies where sea urchins were removed there has been an increase the diversity of algae (Vadas 1977, Keats 1990, Himmelman 1983). However, Lubchenco, 1978) proposed that if herbivores prefer the competitively dominant species, in our case *S.vestitum,* at very low and very high densities diversity would be reduced. In this study, the number of species increased in all plots where densities of sea urchins were reduced densities or removed completely, as foliose algae and several filamentous species coexisted with the crustose algae. However, this may have changed over a longer time scale if a single species of foliose algae was able to outcompete other species when sea urchins were completely removed.

Our results suggest that low densities of sea urchins may affect successional assemblages by delaying or possibly halting the successional process. In previous studies, the trend has been for filamentous red algae to initially increase in cover then begin to decrease in plots where all urchins had been removed, and this has been attributed to the growth of foliose species. However, this early successional species persists where *Centrostephanus* is at intermediate densities, suggesting that it may require some grazing activity to maintain it in the long term. Succession is also time and location specific and can be affected by factors such as the availability of propagules for recruitment. In our study *S. vestitum* was found in all reduced density treatments after 8 weeks. Most *Sargassum* species recruit during summer with ≈90% of released spores settling within 1m of the parent plant and in greater densities to areas that have been cleared of their canopy (Kendrick 1994), a condition that would have been satisfied in the Barrens. It is likely then, that the successional community observed in the Barrens is dependent on the properties of the algae which recruit as well as the grazing ability of sea urchins.

Transition from Fringe to Barrens

Our results indicate that 250% of the natural density of sea urchins in Barrens was required to significantly reduce the total cover of algae and the percent cover of the dominant alga, *Sargassum vestitum*. Additionally, impacts were not immediate and only affected the composition of the entire assemblage after 19 weeks. This study is one of the few that has introduced urchins into an area rich in macroalgae and the only to do so at a range of densities. Andrew, 1993) found that *Centrostephanus* recruits to and survives in both Barrens and Fringe where shelter is present, and so the availability of shelter is thought to be an important factor in the creation of Barrens. However, our study shows that the transformation of an area dominated by macroalgae into Barrens will not occur if the recruitment of sea urchins is below a certain density, which in Fringe dominated by *S. vetitum* during winter must exceed the natural density of the Barrens. Furthermore, grazing did not appear to be linearly related to density, a result that supports Andrew's, 1993) work and suggests the existence of a threshold density. The concept of threshold densities is not uncommon. For example, Wright and Steinberg (2001) demonstrated that densities

of the sea urchin *Heliocedaris eyrthogramma* >80/m^2 dramatically increased the mortality of the red alga *Delisea pulchra*.

It has been hypothesised that as the density of sea urchins increases, the contribution of individual sea urchins to grazing should decrease (Andrew 1993). In the Fringe experiment the area grazed when sea urchins were kept at 50% and 100% of the natural density of Barrens was similar, which suggests that a single urchin can graze twice as much as an individual in a situation where density has been doubled. When *Centrostephanus* was kept at 250% natural density each sea urchin grazed approximately the same area as individuals kept at densities equal to natural densities in Barrens. Therefore, it is possible that at low densities, individual sea urchins contribute more to the area grazed than at natural or higher than natural densities, thus providing some evidence that the foraging behaviour of individual sea urchins changes with density.

Limitations of the study

The use of cages in manipulative experiments is often criticised because of difficulties in interpreting results due to artefacts of the cages themselves (Andrew 1993) such as a reduction in water flow or entrapment of sediment (Kennelly 1991). In this study the effects of caging were controlled for with uncaged plots and plots where sea urchins were kept at natural densities in Fringe and Barrens. In both experiments caging artefacts were minimal as there were rarely significant differences in the cover of dominant species between caged and uncaged control treatments. There was a trend for caged control plots in the Fringe to contain a greater cover of *S. vestitum* than uncaged control plots but this only makes the impact of sea urchin grazing more apparent.

This experiment was designed on a small spatial and temporal scale. At the time of sampling in the Fringe *Sargassum* was blooming, and by the conclusion of the experiment had formed a dense canopy approximately 70 cm high. Different results may have been obtained if the experiment was run during late spring, early summer when *Sargassum* begins to die off (Kennelly 1992), and its biomass is not so overwhelming. Similarly, *Centrostephanus* grazing increases during summer.

Stability of Barrens and Fringe habitats

Once created, Barrens are considered stable (Harrold 1985), generally only reversible by catastrophic events that cause mass mortalities of sea urchins such as influxes of freshwater (Andrew 1991) and disease, or the re-introduction of natural predators such as the sea otter (Estes 1974, Estes 1995, Elner 1990). Consistent with other Australian studies (Andrew 1993, Andrew 1993), this study has found that only a low density (33% of natural density) of sea urchins is needed to maintain Barrens areas relatively free of foliose algae for at least 4 months, indicating Barrens in Australia are probably very stable.

In Fringe, significant differences in the total cover of algae only occurred after 9 weeks, and only when density was increased to 250% of natural density in Barrens, but even then the composition of the entire algal assemblage only differed after 19 weeks of grazing at this density. This indicates that forests where *S. vestitum* dominates are relatively resistant to sea urchin grazing during autumn and winter, and supports previous work (Andrew 1994) which suggests that kelp forests in Australia are also relatively stable. In contrast, studies from the Northern hemisphere have shown kelp forests to be less stable than Barrens. This is largely due to species such as *S. droebachiensis* which have the ability to switch feeding modes from passively grazing in the Barrens to actively invading kelp forests. This depends on a complex interaction between sea urchin density, season and the abundance of potential predators such as lobsters (Bernstein 1983).

Whilst both habitats appear stable, there is a disparity between the density of sea urchins required to maintain and create Barrens. In the Barrens sea urchins are able to regulate the abundance of

foliose algae by consuming the low number of new recruits. In the Fringe habitat it is more difficult for sea urchins to have a significant impact simply because of the large biomass of algae. In addition, many species of algae, such as kelps, have short dispersal distances (Kennelly 1987) and so that the Fringe always has a potential supply of recruits.

Importance of feeding preferences

Many North American species such as *Strongylocentotus franciscanus* and *S. droebachenisis* show preferences both in the laboratory and the field (Himmelman 1990, Vadas 1977). The strength of these preference decreases as food availability decreases and as a result are irrelevant in most natural situations (Vadas 1977). In contrast, in New Zealand the preference of *Evechinus chloroticus* for *Ecklonia radiata* over other species in the Fringe is ecologically significant as it results in the differential loss of algae from natural stands (Schiel 1982). In our feeding trials *Censtrostephanus* preferred *S. linearfolium, Z. diesingiana* and *E. radiata* over *S. vestitum* but in caging experiments, *Centrostephanus* grazing had a significant impact on the cover of the most abundant alga, *S. vestitum,* but not on the next most abundant species, *S. linearfolium* or on the cover of *Z. diesingiana.* Furthermore, Steinberg, 1995) found that *Centrostephanus* did not significantly avoid *S. vestitum* when presented with several types of algae in the Barrens. Perhaps these conflicting results can be attributed to the methods used in the feeding assays as the sea urchins in our study were not food limited (starved) and therefore more likely to exhibit preferences. In any case, these inconsistencies highlight the difficulty in extrapolating the results of controlled feeding experiments to natural situations. These difficulties exist because assays have the potential to overestimate the impact of preferences on algal assemblages (Reusink 2000). Although our assay was conducted in the field it was within an artificial environment; sea urchins grazed over an area much smaller than their natural foraging ranges, we did not account for intra-specific interactions and plant fragments were used which may illicit different responses to whole plants (Reusink 2000).

Alternatively, ecological constraints, such as the spatial distribution of plants, have been used to explain the preferences of terrestrial herbivores in the field. Preference for an alga may be inhibited if the species is rare or the animal is not sufficiently mobile. However, *Centrostephanus* is capable of travelling up to 5m in one night and the cover of *S. linearifolium,* a more preferred alga in trials, was around 20% in all plots and so cannot be considered rare. Conversely, the cover of *S. vestitum* was dense making it likely to be encountered first and possibly inhibit *Centrostephanus'* ability to locate and consume *S. linearfolium.* Therefore, the feeding preferences of *Centrostephanus* have the potential to be ecologically significant, but when faced with a situation, as in the Fringe experiment, where one species of algae dominates they are largely unimportant.

Implications for management of the fishery for Centrostephanus

Due to the limited availability of food in the Barrens that results in poor quality roe (Blount 2002, Byrne 1998). The fishery for *Centrostephanus* in NSW is small (<50 t yr^{-1}), due to variability in the quality of roe amongst other things. Currently, two methods for improving the roe quality of *Centrostephanus* in the wild on a commercial scale are being investigated: reducing densities of individuals in Barrens and transplanting individuals to Fringe.

Blount *et al* (2002) has demonstrated that a reduction in sea urchin density to 33% natural density can result in a two fold increase in the yield of roe, an improvement of gonad colour and recruitment may also be increased locally. Our experiment indicates that a reduction of this magnitude for five months may maintain the Barrens relatively free of foliose algae but may promote an increase in filamentous red algal cover. Reductions in density caused by harvesting may have consequences for invertebrate communities, detrimentally effecting the abundance of

limpets (Andrew 1993, Fletcher 1987) and other invertebrates such as shrimp and juvenile gastropods, which live underneath the sea urchins (Davis 2002).

Our study has demonstrated the Fringe areas have the capacity to support densities of sea urchins similar to the natural densities in the Barrens for at least four months without changing the assemblage of algae. The effects of *S. vestitum,* not a preferred alga in the feeding trial, on roe quality are unknown but need to be considered as it is a dominant alga in many Fringe habitats in Sydney. Noting the temporal and scale limitations of this study both of the proposed methods for enhancing roe may be viable as the impacts of manipulating sea urchin density in the Barrens and Fringe on existing algal assemblages appears to be minimal. However, the sustainability of this method on a larger scale will depend on the time frame required to improve roe quality and possibly also on season.

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APPENDIX 5:

Patterns in the distribution and abundance of fauna associated with the sea urchin *Centrostephanus rodgersii* (Agassiz) in New South Wales, Australia

(submitted to *Mar. Freshwater. Res*.)

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Abstract

Boreholes of the sea urchin *Centrostephanus rodgersii* (Agassiz) provide a microhabitat for a diverse assemblage of macrofauna on New South Wales' subtidal reefs. The aim of this study was to quantify patterns in the distribution and abundance of fauna associated with *C. rodgersii* at a variety of spatial and temporal scales, and between habitats, since this has not been done previously and because there is potential for the fishery for *C. rodgersii* to expand. Natural patterns in abundance were investigated at four locations along the coast of New South Wales. Over 100 taxa were sampled, 53% of which were represented by <10 individuals across all samples. Where present, variations in abundance of individual species occurred between sites within locations and only one species exhibited significant differences in abundance between the fringe and barrens habitats. Abundances of individual taxa changed in different ways at two sites over a period of 10 months. The composition of whole assemblages under urchins differed between sites, locations and habitats and also changed over time. Several models that may explain this variation and are discussed, as well as the implications for detecting changes related to fishing.

Introduction

Temperate areas in Australia's marine environment are diverse in their assemblages of plants and animals, with many endemic fishes (85% endemic), molluscs (95%), red algae (75%) and echinoderms (90%) (Poore 1995). Absolute numbers of species are not known for many marine environments, including rocky subtidal reefs, and trends in species diversity are poorly documented along Australia's coastline (Poore 1995; O'Hara 2001). Studies of the patterns of distribution and abundance of benthic marine fauna in eastern Australia have concentrated in the intertidal (e.g. Fairweather 1988; Underwood and Verstegen, 1988; Worthington and Fairweather 1989; Underwood and Chapman 1992; Peake and Quinn 1993; Chapman 1994; Underwood and Chapman 1996; Beck 1998; Underwood and Chapman 1998a, b). Fewer studies have been made in subtidal reef habitats and consequently our knowledge of these areas is considerably less (Edgar 1984; Underwood and Kennelly 1990; Underwood *et al.* 1991; Keough and Butler 1995; Smith *et al.* 1996; O'Hara 2001). Studies on the fauna of subtidal reefs have been largely restricted to commercially important species such as abalone (Shepherd 1973; Andrew and Underwood 1992;

Andrew *et al.* 1998), sessile species (Butler and Chesson 1990; Davis and Ward 1999), or large abundant species such as some sea urchins and fishes (Fletcher 1987; Andrew and Underwood 1989; Jones and Andrew 1990; Andrew and Underwood 1993; Andrew 1993).

Centrostephanus rodgersii (Agassiz) is a diadematid sea urchin that is abundant on subtidal reefs of southeastern Australia (Andrew and Underwood 1989; Andrew and O'Neill 2000). Grazing by *C. rodgersii* is an important determinant of the biodiversity of subtidal rocky reefs, by maintaining the crustose coralline-dominated 'barrens' habitat and the associated assemblages (Fletcher 1987; Andrew and Underwood 1989). During daylight hours, *C. rodgersii* inhabit crevices in bedrock and around boulders, and boreholes that have been gradually excavated by scraping into the rocky substratum. During daylight hours, the presence of *C. rodgersii* may act as a biogenic habitat that may be important to the ecology of subtidal reefs in New South Wales by contributing to reef habitat heterogeneity and providing shelter for a diverse range of small benthic animals. Other biogenic habitats such as sponges and coral heads are known to support unique and diverse faunas (Abele and Patton 1976; Westinga and Hoetjes 1981; Theil and Vásquez 2000). Urchins may be important to associated fauna for a number of reasons such as the provision of refuge or food resources. Therefore the distribution and abundance of *C. rodgersii* may have a large influence on the abundance and distribution of fauna that shelter beneath them. The importance of *C. rodgersii* as a biogenic habitat will depend on the ecological requirements and status of species that associate with it, including whether these associated species are able to utilise other habitats and whether they are common or rare on the reef..

The commercial catch of *C. rodgersii* in New South Wales has increased in recent years, and the fishery has potential to expand by an order of magnitude or more and this may lead to impacts upon other species in subtidal habitats. Fishing could potentially alter the macrofaunal assemblages inhabiting *C. rodgersii*, particularly if these assemblages depend on *C rodgersii* for habitat. Fishing indirectly affects other species in a system and it is appropriate to consider the multi-species aspects of exploitation.

Understanding changes in marine assemblages in response to anthropogenic activities requires a good understanding of their natural spatial and temporal variation (Underwood 1991; Underwood and Chapman 1998a). In order to document natural variation adequately, several spatial and temporal scales should be sampled. For example, patchiness in microhabitats, or resources, can influence densities of organisms at small-scales (Morrisey *et al.* 1992; Edgar and Barrett 2002) whereas variations in the physical environment may operate at larger scales (Edgar *et al.* 1997, 1999, 2000). Some processes can operate at more than one scale. Sampling at a hierarchy of scales when documenting natural variation in abundance is important because it identifies the scale at which processes are likely to be operating (Underwood 1994). Processes can only be confirmed after generating models to explain variation at different scales and testing them via manipulative experiments (Chapman 1994; Underwood 1997; Underwood and Chapman 1998a, b).

This study aimed to identify the species associated with *C. rodgersii* in New South Wales and to quantitatively examine their spatial and temporal patterns in abundance. This forms an initial step towards understanding the potential impacts of removing *C. rodgersii* from subtidal reefs as a consequence of fishing. Specifically, we tested the following null hypotheses relating to the macrofauna associated with *C. rodgersii*:, 1) macrofaunal assemblages and abundances of individual taxa do not differ between locations along the New South Wales coast separated by 100s of km and between sites within locations separated by ~2 km; (2) macrofaunal assemblages and abundances of individual taxa in fringe and barrens habitats do not differ; and (3) macrofaunal assemblages and abundances of individual taxa do not vary significantly over a period of one year, and this pattern is consistent across sites separated by ~2 km.

Methods

Study locations and habitats

In order to lend some generality to the study, it was important that the extent of sampling spanned the distribution of *C. rodgersii* in New South Wales. Therefore, four locations spanning 400 km of the New South Wales coastline were sampled: Nelson Bay, Cronulla, Ulladulla and Eden (Fig. 1). *C. rodgersii* is very common in two habitats on rocky reefs (fringe and barrens, *sensu* Underwood *et al.* 1991). However, urchins from both habitats were sampled because of the need to establish whether assemblages under urchins in the fringe are unique, and also because the fishery may also target *C rodgersii* in the barrens at some time in the future if the roe from those individuals can be enhanced.

Sampling design

A pilot study conducted at Cronulla and Eden, in September and October of 2000, was used as a guide for the design of the main study. Variation in the abundance of individual taxa did not differ between plots ($25m^2$) nested within sites and so this spatial scale was not included for the main study. Considerable variation was found between individual replicate samples during the pilot study. Components of variance analysis and optimal replication calculations showed that large numbers of replicates (>50) would be needed to satisfy an acceptable level of precision. Sampling many replicates is both time and cost prohibitive and can be potentially destructive to populations with apparent low abundances. These constraints were considered with a cost-benefit analysis producing the most cost-effective design for the main study.

A nested sampling design was employed for this study to enable the abundance and distribution of fauna associated with *C. rodgersii* to be estimated at a number of spatial scales. Within each location, three sites separated by approximately 2 km were sampled (Fig. 1) by collecting fauna from six replicate urchins in each of the fringe and barrens habitats. A total of 144 urchin samples were collected. Sampling was carried out in December 2000 and all locations were sampled within a ten-day period.

Temporal variation was assessed at one location (Cronulla) in the fringe habitat, from samples collected in September 2000, December 2000, 27 June 2001, 28 June 2001, and July 2001. Six urchins were sampled on each occasion from two sites, approximately 2 km apart.

Field and laboratory procedures

Divers on SCUBA using a suction sampling device known as an “air lift” with a 180 μm mesh bag attached sampled mobile fauna beneath *C. rodgersii*. Urchins were selected haphazardly in the fringe and barrens habitats. Fringe habitat was determined by depth and the presence of geniculate coralline algae and the foliose algae *Ecklonia radiata and Sargassum sp*. Fauna were collected from below solitary urchins, that is, those whose shelter was a single eroded hollow in the rock substrate. Urchins of ~ 100 mm test diameter were sampled. By selecting solitary urchins of a similar size, the area of substrate sampled was consistent among replicates since the size of the urchin dictates the size of the borehole.

Each urchin was removed carefully with a hook and the mobile fauna residing underneath and within the spines of the urchin was quickly suctioned into the collection bag. The identity and number of escaping fauna (very few) were recorded on a slate. After suctioning, the plankton mesh bag was removed, sealed and stored. New plankton bags were fitted for each subsequent sample. Urchins were returned to their boreholes following sampling.

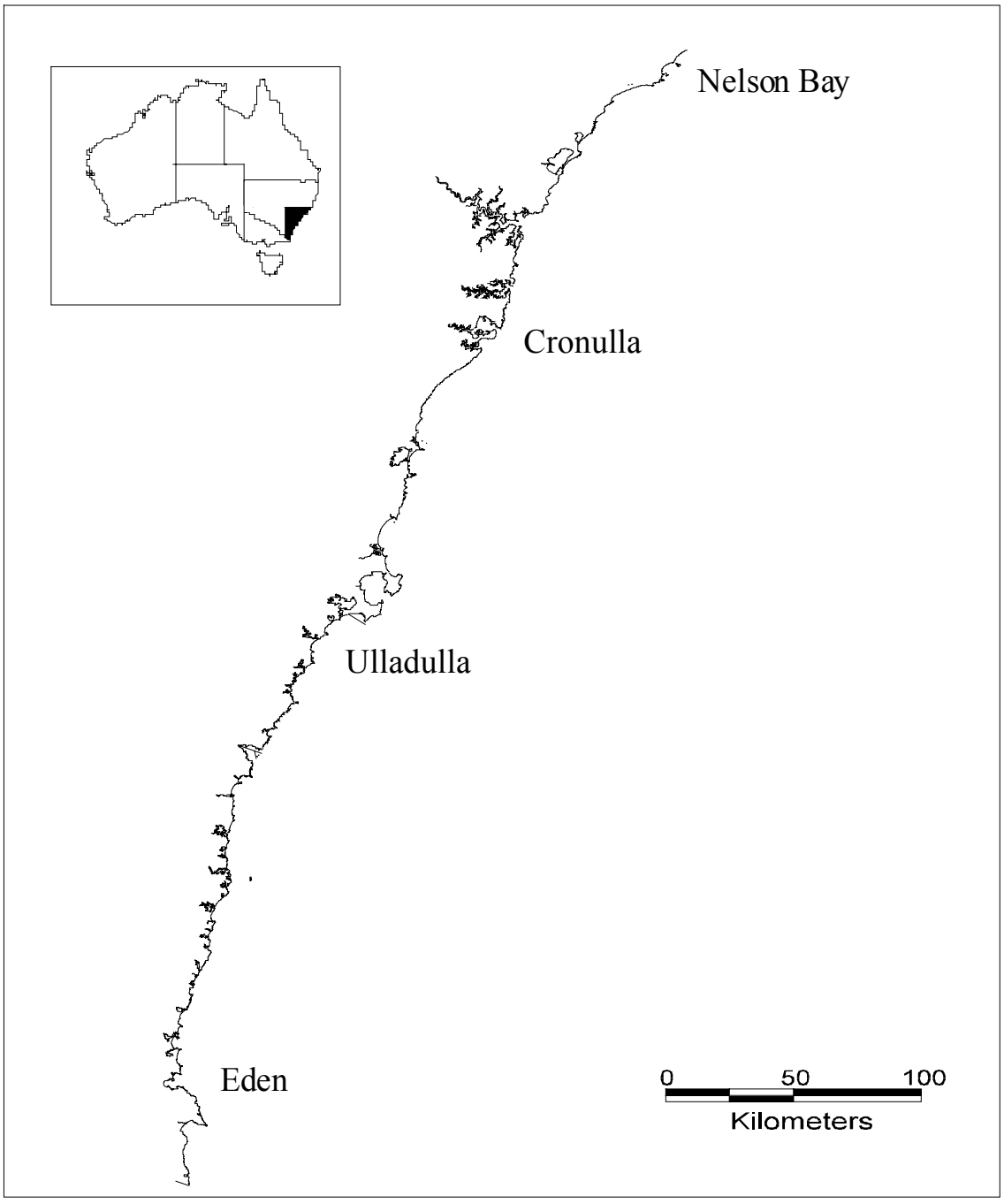

Figure 1. Position of study locations.

Samples of macrofauna were fixed in 5% formalin in seawater solution and returned to the laboratory for sorting, identifying and enumeration. Samples were washed through a 0.5 mm sieve, sorted under a magnifying lamp and stereo microscope, and specimens identified. Specimens were identified to the highest possible taxonomic resolution, with the exception of amphipods and polychaetes which were identified to class level only. Specimens unable to be identified were classified as different 'morphospecies' (Oliver and Beattie 1996) within classes.

Univariate statistical analyses

Taxa represented by at least 50 individuals, and taxa that were of special interest, were chosen for analysis of variance (ANOVA). The latter included taxa that, based on the literature, were likely to have a strong association with *C. rodgersii*. For example, the purple shrimp *Athanas granti* (Alpheidae) belongs to the same genus as another species (*Athanas indicus*) that is known to be associated with the sea urchin *Echinometra mathaei* at the Gulf of Elat in the Red Sea (Pomeranz and Tsurnamal 1976; Gherardi 1991). Taxonomic richness, Shannon-Weiner diversity, total numbers of individuals, and 13 selected taxa were analysed for spatial patterns. Taxonomic richness, Shannon-Weiner diversity, total numbers of individuals, and 11 taxa were tested for temporal changes. Any significant differences were further investigated using Student-Newman-Keuls (SNK) tests to identify differences between pairs of means (Underwood 1981).

For the spatial comparisons the factor of Location was treated as random and orthogonal, Habitat as fixed and orthogonal and Sites as random and nested in Location and Habitat. For the temporal comparisons the factors Time and Site were treated as random and orthogonal. Data were tested for homogeneity of variances using Cochran's Test prior to analysis and wherever necessary data were transformed using the Ln (x +1) transformation to homogenise variances (Underwood 1981). In some instances, transformations of the data were unsuccessful and analyses were done using untransformed data and an adjusted significance level of $\alpha = 0.01$ (Underwood 1981). Where they occurred, significant interactions meant that main effects could not be interpreted and these were not reported in ANOVA tables (Underwood 1981).

Multivariate analyses

Analyses were done using Bray-Curtis similarity matrices in which raw data were fourth-root transformed because some taxa were very abundant, 100's of individuals) in some samples. Differences between assemblages were visualized by non-metric multidimensional scaling (nMDS) ordinations using PRIMER software (Clarke 1993). Stress values <0.15 indicate that the ordinations were reliable representations of the true relationships between samples (Clarke 1993). Ordinations were based on site-average data because the large number of samples produced high stress values, which precluded meaningful interpretation of relationships between assemblages. Two-way nested non-parametric MANOVA (Anderson 2001) using permutation of residuals (full model) and 4999 permutations was used to test the null hypotheses that assemblages did not differ between locations or between sites nested within locations, and that assemblages did not differ between habitats at each location. The hypothesis that assemblage composition was consistent through time was tested by two-way crossed non-parametric MANOVA (Anderson 2001) using permutation of residuals (reduced model) and 4999 permutations; times and sites were analysed as random factors. Species contributing to similarity/dissimilarity between assemblages were determined by the SIMPER routine in PRIMER. The average percentage contributions that each species made to dissimilarity were calculated to determine which species were most important in distinguishing between sites, locations and habitats.

Results

General findings

A total of 100 taxa (over 10,000 individuals) were recorded from boreholes below *C. rodgersii* over the course of this study. Spatial comparisons of four locations along New South Wales in December 2000 collected 8030 individuals representing 73 taxa. Seventy-three taxa were found in the fringe habitat and 47 taxa were found in the barrens habitat. All taxa found in the barrens were also present in the fringe. Of the 73 taxa identified (amphipods and worms excluded) nearly 50% were gastropods and 15% were crustaceans.

Most taxa were present in low numbers: 53% of taxa were represented by less than 10 individuals across all samples; 30% of taxa were represented by 11-50 individuals; 5% of taxa were represented by 51-100 individuals; and 12% of taxa were represented by more than 100 individuals. Amphipods, polychaetes and the hermit crab (*Pagurus lacertosus*) were usually the most abundant taxa. On average 55.8 ± 8.1 (mean ± standard error) individuals and 7.8 ± 0.3 taxa were found under each urchin (n = 144 urchins).

Spatial comparisons - Univariate analyses

Different spatial patterns of abundance were found among the variables tested. Taxonomic richness only varied between sites at one location (Cronulla) and total abundance of macrofauna did not differ at any of the spatial scales examined (Table 1; Fig. 2). Shannon diversity showed a significant interaction between site (location) and habitat.

For those taxa where main effects could be interpreted, none showed significant differences in abundance at the scale of locations (Table 1; Fig. 2). Most variation in abundance occurred between sites (location). Site-level variation was the only pattern observed for seven taxa: *Rhynchocinetes serratus* (hingebeak shrimp); *Agnewia tritoniformis* (whelk); the gastropods *Rissoina fasciata* and *Anabathron lene*; polychaetes, *Apasmogaster costatus* (clingfish) and *Athanas granti* (purple shrimp) (Table 1; Fig. 2). Post-hoc comparisons of means showed that significant differences between sites for these taxa were not widespread and were only evident in one or two of the four locations, and the locations where these differences occurred varied for different taxa. Amphipod abundance varied with both location and habitat: amphipods were more abundant in the fringe habitat at the northern location of Nelson Bay and were more abundant in the barrens in the south at Eden (Table 1; Fig. 2).

Abundance of only one species, the hermit crab *Pagurus lacertosus*, differed significantly between fringe and barrens habitats (Table 1; Fig. 2): *P. lacertosus* were more abundant in the fringe habitat. The brittlestar *Clarkoma pulcra* did not occur in both habitats in all locations (Table 1; Fig. 2): it was not found in barrens in Ulladulla and Eden and when it did occur in both habitats, abundances were higher in the fringe.

Table 1. Summary of ANOVA results for spatial patterns in abundance of macrofauna from fringe and barrens habitats in December 2000 (n=6). In this and following tables *C* = Cochrans *C*-value; ns = *P*>0.05; * = *P*<0.05; ** = *P*<0.01; *** = *P*<0.001.

		Diversity *C* = 0.13ns		Richness *C* = 0.11ns		Total abundance *C*= 0.6448**		*Pagurus lacertosus* *C* = 0.11ns	
Source of variation	df	MS	*F*	MS	*F*	MS	*F*	MS	*F*
Location	3	4.34		2.40	2.94	14796.9	1.4	2.91	0.65
Site (Location)	8	0.66		0.89	2.41*	10574.0	1.27	4.51	3.30**
Habitat	1	0.12		3.09	5.62	43611.3	1.62	61.23	29.48*
Location x Habitat	3	0.44		0.55	1.38	26838.3	2.45	2.08	1.89
Habitat x Site (Loc)	8	0.57	2.58*	0.40	1.17	10659.4	1.32	1.10	0.81
Residuals	12	0.22		0.34		8321.18		1.37	
Total	14								
		C. pulcra *C* = 0.23**		*R. serratus* *C = 0.11ns*		*A. tritoniformis* *C = 0.13ns*		*R. fasciata* *C* = 0.54**	
Source of variation	df	MS	*F*	MS	*F*	MS	*F*	MS	*F*
Location	3	0.32		1.37	0.76	1.38	1.28	33.01	0.92
Site (Location)	8	0.11		1.80	4.69**	1.09	3.73**	35.76	2.68*
Habitat	1	0.84		2.04	2.15	0.03	0.06	13.44	0.49
Location x Habitat	3	0.41	9.94**	0.95	1.79	0.47	1.7	27.46	1.61
Habitat x Site (Loc)	8	0.04	0.33	0.53	1.38	0.28	0.96	17.10	1.28
Residuals	120	0.13		0.38		0.29		13.33	
Total	143								
		A. lene *C = 0.95***		Polychaetes C= 0.2358**		*A. costatus* *C = 0.11ns*		*A. granti* *C= 0.12ns*	
Source of variation	df	MS	*F*	MS	*F*	MS	*F*	MS	*F*
Location	3	232.91	1.67	47.8	2.18	0.07	0.23	0.36	0.61
Site (Location)	8	139.48	3.08**	21.9	2.76**	0.32	2.63*	0.58	2.90*
Habitat	1	112.01	1.24	0.69	0.44	0.01	0.01	0.47	1.03
Location x Habitat	3	90.17	1.85	1.56	0.18	0.60	2.98	0.45	2.27
Habitat x Site (Loc)	8	48.84	1.08	8.62	1.08	0.20	1.68	0.20	1.00
Residuals	120	45.22		7.96		0.12		0.20	
Total	143								
		C. brunneus *C = 0.87***		Amphipods C = 0.12ns		*C. clangulus* *C = 0.3317***		*P. mufria* *C = 0.4325***	
Source of variation	df	MS	*F*	MS	*F*	MS	*F*	MS	*F*
Location	3	115.06		22.86		43.23	5.24	5.82	3.79
Site (Location)	8	135.51		6.56		8.26	2.25	1.53	1.11
Habitat	1	186.78		0.01		11.67	1.59	4.69	7.8
Location x Habitat	3	114.74		19.76	12.82*	7.36	0.92	0.60	0.47
Habitat x Site (Loc)	8	127.42	3.15**	1.54	1.30	7.98	2.18	1.27	0.92
Residuals	120	40.43		1.19		3.67		1.38	
Total	143								

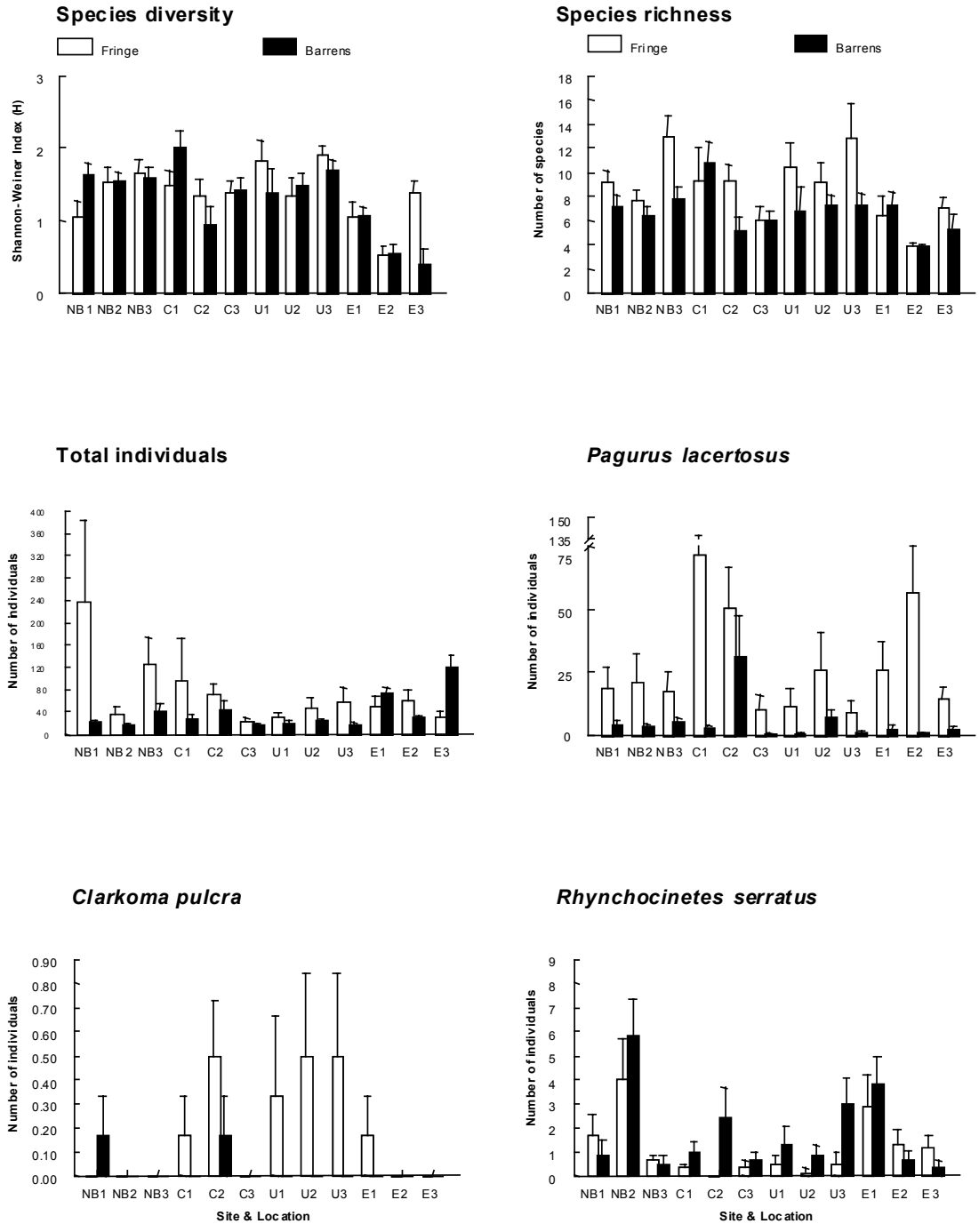

Figure 2. Diversity of taxa, richness of taxa, total abundance and abundance of 13 taxa in each of three sites per location in fringe and barrens habitats in December 2000. Values shown are mean ± SE (n=6). NB = Nelson Bay; C = Cronulla; U = Ulladulla; E = Eden.

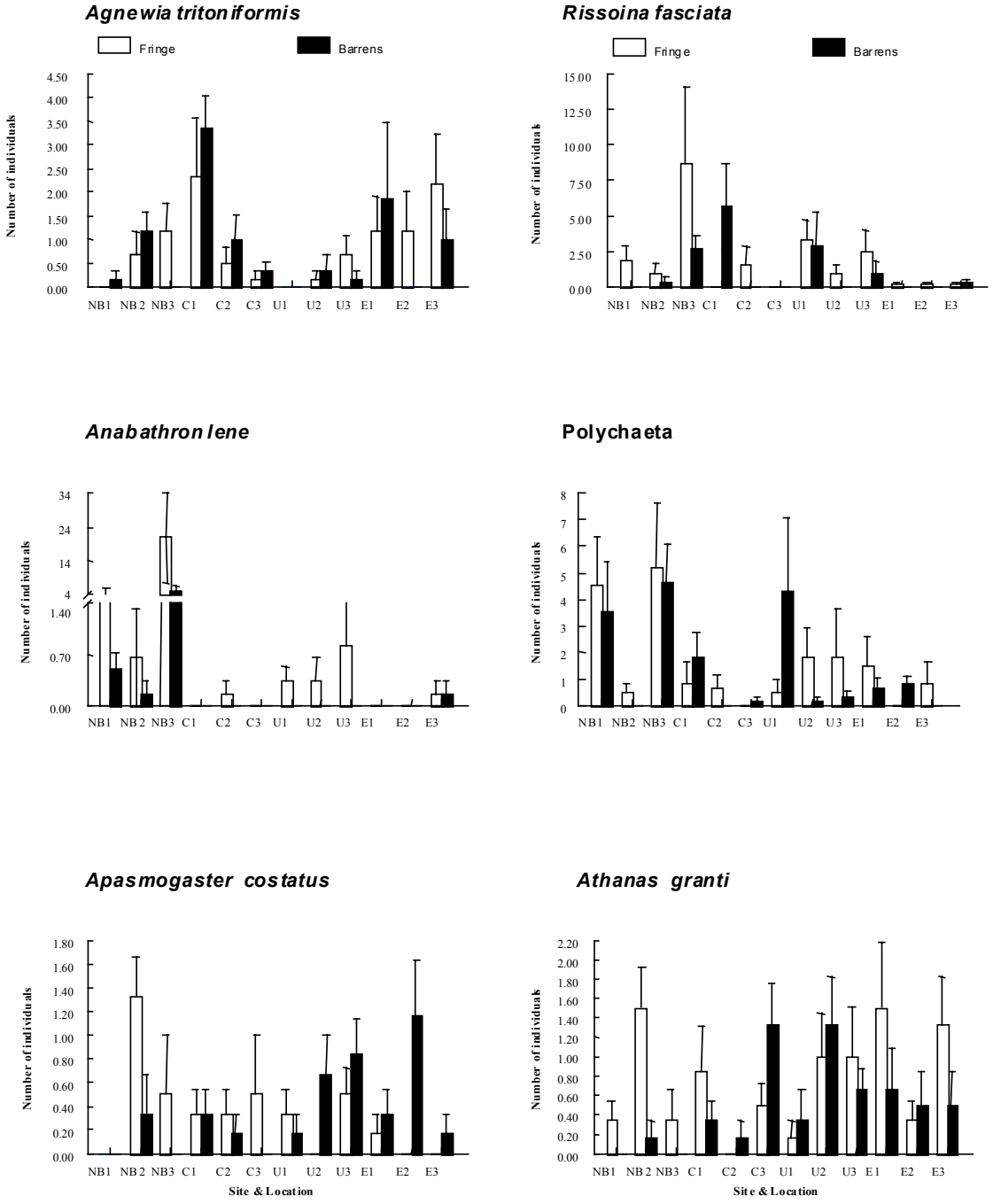

Figure 2 (cont.) Diversity of taxa, richness of taxa, total abundance and abundance of 13 taxa in each of three sites per location in fringe and barrens habitats in December 2000. Values shown are mean ± SE (n=6). NB = Nelson Bay; C = Cronulla; U = Ulladulla; E = Eden.

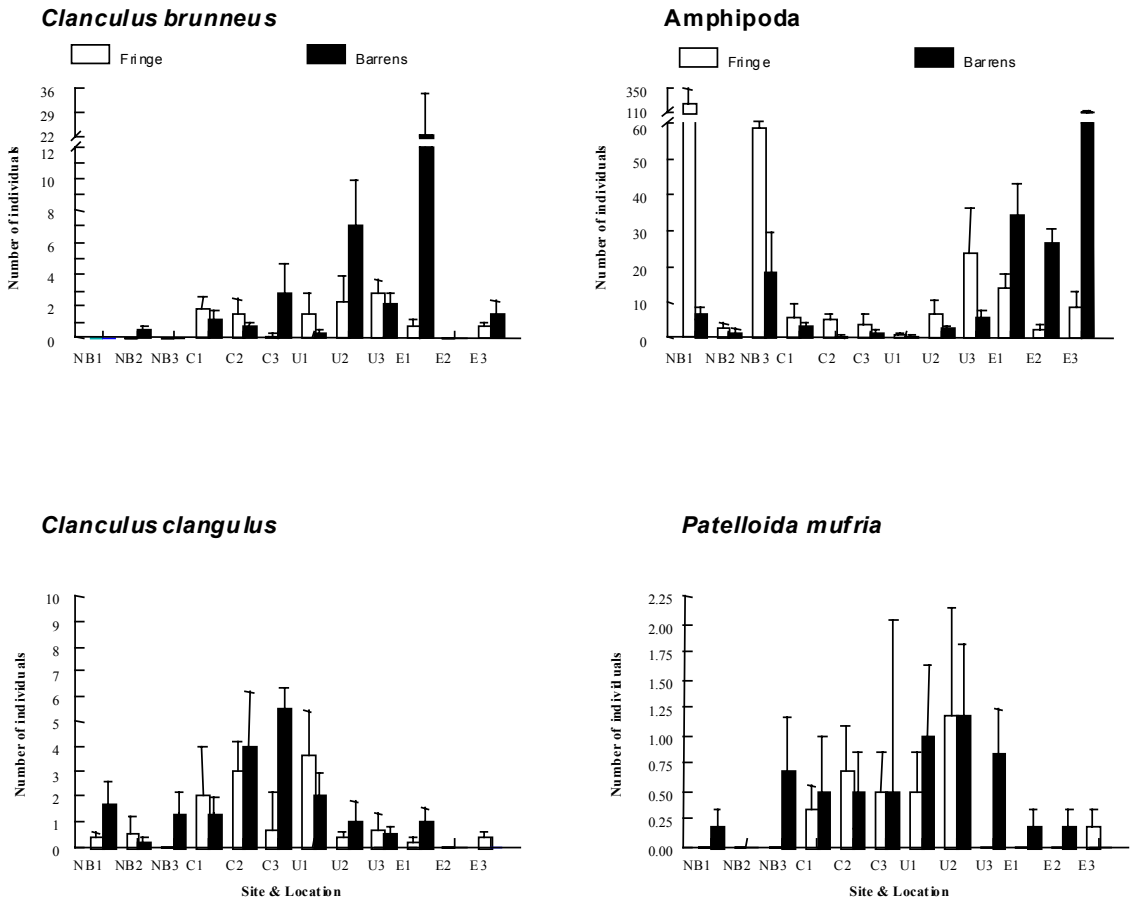

Figure 2. (cont.) Diversity of taxa, richness of taxa, total abundance and abundance of 13 taxa in each of three sites per location in fringe and barrens habitats in December 2000. Values shown are mean ± SE (n=6). NB = Nelson Bay; C = Cronulla; U = Ulladulla; E = Eden.

Temporal variation

Diversity, taxonomic richness, and abundances of the gastropod *Pisinna* sp. and the brittlestar *Clarkoma pulcra* varied through time, but in different ways in each site (Table 2; Fig. 3). For example the abundance of *Pisinna* sp. increased over time, however this trend was only evident at one site. The brittlestar *Clarkoma pulcra* varied in abundance through time, but was only present at one site on most sampling occasions. The purple shrimp *Athanas granti* varied in abundance through time and was not detected at one site on three out of five sampling occasions. Total abundance of macrofauna and abundance of the hermit crab *Pagurus lacertosus* did not change through time, although there were consistent differences between sites (Table 2; Fig. 3). Seven of the 11 taxa tested did not vary significantly in abundance through time or between sites at Cronulla (Table 2; Fig. 3).

Table 2. Summary of ANOVA results for temporal patterns in abundance of macrofauna in the fringe habitat at Cronulla (two nested sites) between September 2000 and July 2001 (n=6).

		Diversity *C* = 0.2114ns		Richness *C* = 0.1598ns		Total abundance *C*= 0.1783ns		*C. pulcra* *C* = 0.2672ns	
Source of variation	df	MS	*F*	MS	*F*	MS	*F*	MS	*F*
Time	4	1.85		0.93		0.61	1.8	1.05	
Site	1	0.01		4.99		39.90	118.0	7.55	
Time x Site	4	1.32	7.91**	0.68	5.56**	0.34	0.47	0.56	2.78*
Residuals	50	0.17		0.12		0.71		0.20	
Total	59								
		A. granti *C*= 0.4305**		*P. lacertosus* *C* = 0.1634ns		*Pisinna sp.* *C* = 0.5102**		*A. incidatus* *C*= 0.4168**	
Source of variation	df	MS	*F*	MS	*F*	MS	*F*	MS	*F*
Time	4	1.96	33.57*	1.39	1.23	23.9		22.35	1.4
Site	1	1.35	23.14*	104.04	92.15**	68.2		74.82	4.68
Time x Site	4	0.06	0.12	1.13	0.82	23.9	4.58**	15.98	3.2
Residuals	50	0.50		1.38		5.23		4.99	
Total	59								
		Amphipods *C* = 0.4021**		*A. costatus* *C* = 0.2960ns		*C. brunneus* *C* = 0.9763**		*C. clangulus* *C* = 0.6648**	
Source of variation	df	MS	*F*	MS	*F*	MS	*F*	MS	*F*
Time	4	50.94	3.44	0.05	0.49	44.32	0.84	80.56	1.35
Site	1	68.27	4.61	0.17	1.66	36.82	0.70	28.02	0.47
Time x Site	4	14.81	1.59	0.10	0.94	52.9	1.93	59.56	2.93
Residuals	50	9.29		0.11		27.37		20.30	
Total	59								
		Foramnifera *C*= 0.3893**		Unknown Gastropod #5 *C*= 0.7059**					
Source of variation	df	MS	*F*	MS	*F*				
Time	4	32.94	2.77	11.02	1.0				
Site	1	32.27	2.71	43.35	3.93				
Time x Site	4	11.89	1.72	11.02	2.15				
Residuals	50	6.93		5.12					
Total	59								

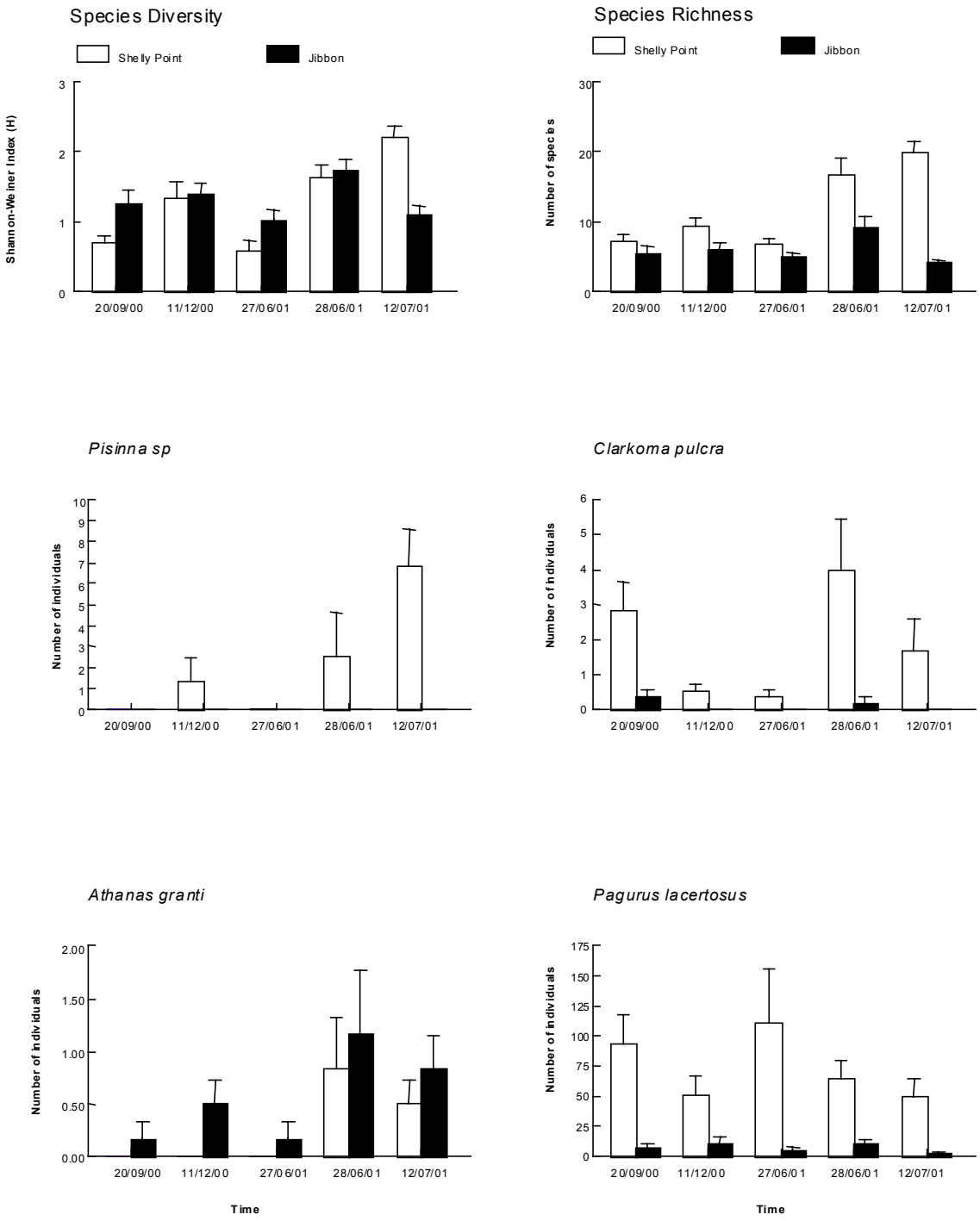

Figure 3. Diversity of taxa, richness of taxa, total abundance and abundance of 11 taxa at two sites in the fringe habitat at Cronulla between September 2000 and July 2001. Values shown are mean ± SE (n=6) for each sampling occasion.

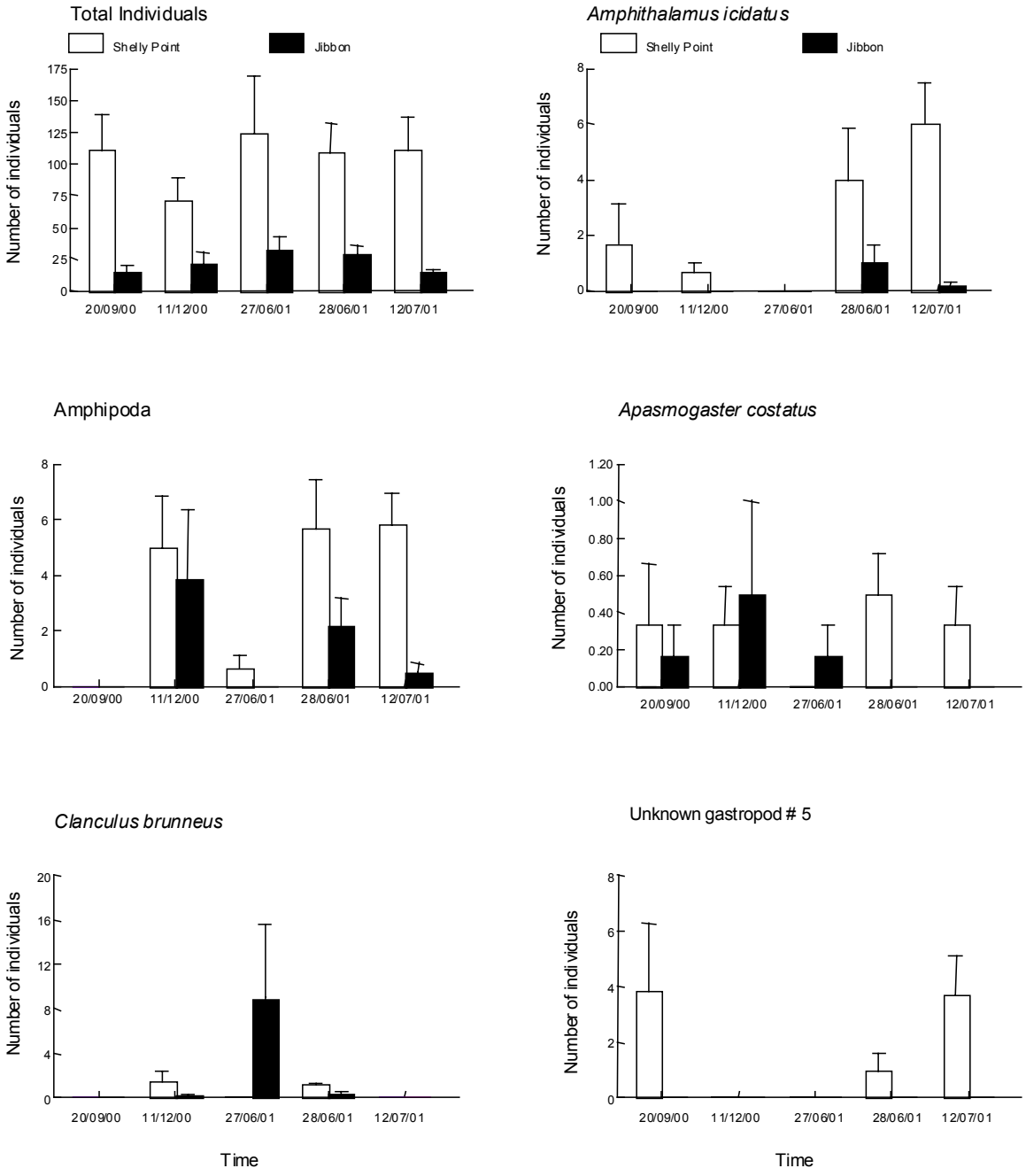

Figure 3. (cont.) Diversity of taxa, richness of taxa, total abundance and abundance of 11 taxa at two sites in the fringe habitat at Cronulla between September 2000 and July 2001. Values shown are mean ± SE (n=6) for each sampling occasion.

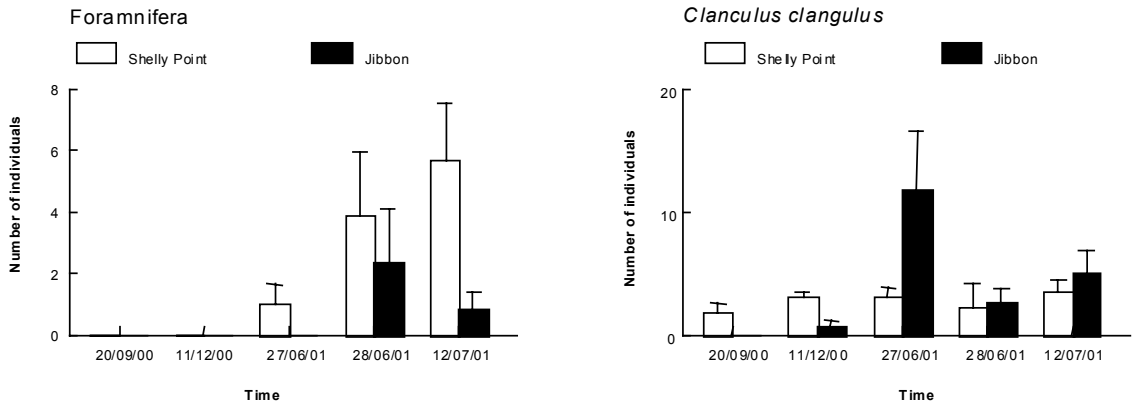

Figure 3. (cont.) Diversity of taxa, richness of taxa, total abundance and abundance of 11 taxa at two sites in the fringe habitat at Cronulla between September 2000 and July 2001. Values shown are mean ± SE (n=6) for each sampling occasion.

Spatial Comparisons - Multivariate analyses

nMDS ordinations for each habitat suggest dissimilarity in assemblage composition between some locations (Fig. 4). This was confirmed by npMANOVA which showed, for both habitats, that locations differed in their assemblages (Table 3). Results of pairwise comparisons of locations differed in each habitat with the exception of Eden which differed from Cronulla and Ulladulla in both habitats (Table 3). Many taxa (49) contributed to the dissimilarity of assemblages between locations with each making small contributions. Taxa contributing most to average dissimilarity included amphipods, 10%), the hermit crab *Pagurus lacertosus* (7.5%) and the hingebeak shrimp *Rhynchocinetes serratus* (5.7%).

The extent of site (location) differences in assemblages varied between habitats (Fig. 4). Assemblages in the fringe habitat differed between some sites in all locations (Table 3). Assemblages in the barrens habitat differed between all sites in each location, with the exception of Ulladulla where sites were not significantly different. Similarity among replicate samples within each site was generally low, ranging from 28% to 51% in the fringe habitat and from 20 to 61% in barrens habitat.

Assemblages differed between fringe and barrens habitats at Ulladulla ($F_{1,4}$ = 2.33, P<0.05) and Eden ($F_{1,4}$ = 3.31, P<0.05), but not at Nelson Bay ($F_{1,4}$ = 0.93) and Cronulla ($F_{1,4}$ = 1.32) (Fig. 5). Thirty-five taxa contributed to dissimilarity of assemblages between habitats, although differences in abundance of a common suite of 5 taxa contributed to most of the dissimilarity (Table 4). In general, patterns of difference between habitats for abundances of these taxa were retained across all locations e.g. the hermit crab *Pagurus lacertosus* was always more abundant in the fringe habitat. However, hingeback shrimp *Rhynchocinetes serratus* were more abundant in the barrens habitat at Nelson Bay and Ulladulla but more abundant in the fringe habitat at Eden. Amphipods were more abundant in the fringe habitat at Nelson Bay, Cronulla and Ulladulla, and more abundant in the barrens habitat at Eden.

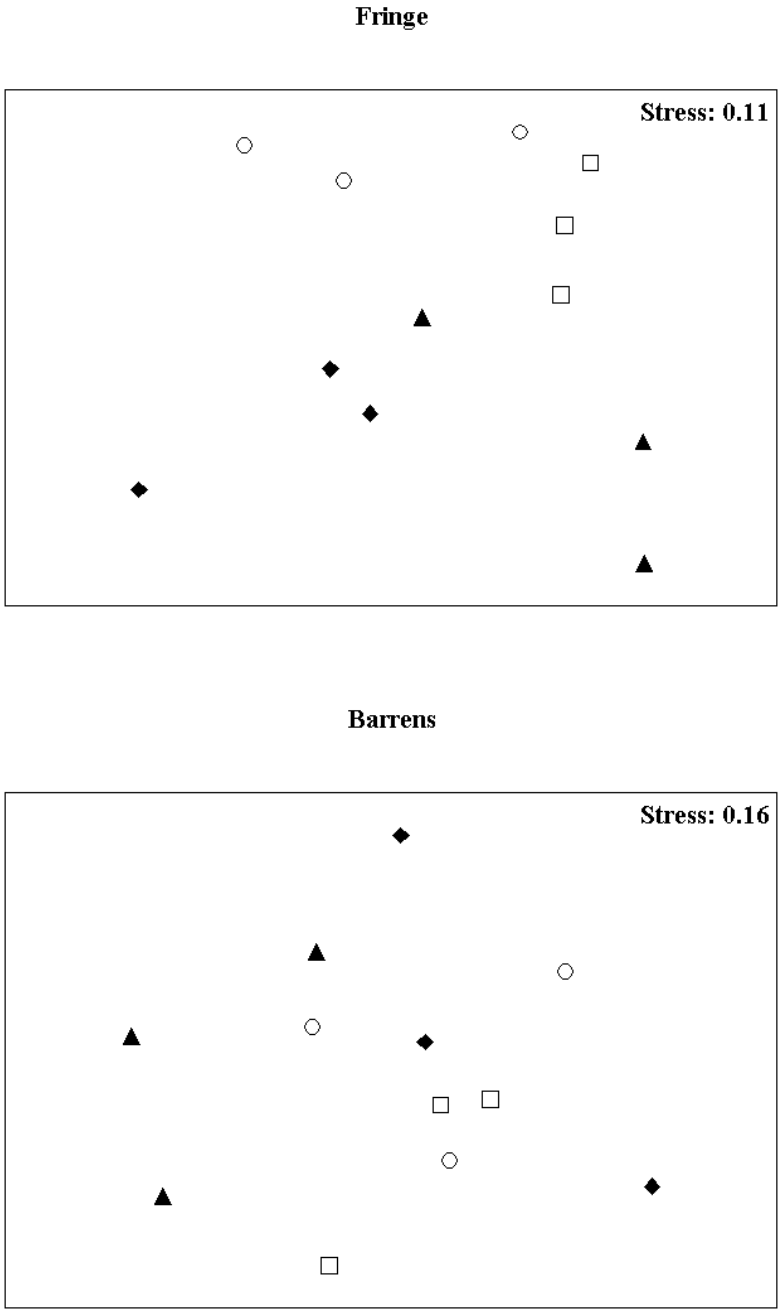

Figure 4. nMDS ordinations of faunal assemblages sampled from beneath *C. rodgersii* in December 2000 from fringe and barrens habitats in four locations: ▲ Nelson Bay; ○ Cronulla; ❑ Ulladulla; ◆ Eden.

Table 3. Summary of non-parametric MANOVA results comparing assemblage composition between locations, and sites within locations, in each habitat.

		Fringe		Barrens	
Source	*df*	MS	*F*	MS	*F*
Location	3	7690.03	2.14**	11328.20	2.23**
Site (Location)	8	3588.89	1.82***	5079.78	3.07***
Residual	60	1967.02		1652.26	

Results of a posteriori comparisons between locations in each habitat, showing *t*-values and significance levels.

Locations	Fringe	Barrens
NB vs CR	1.52 *	1.26 ns
NB vs UL	1.31 ns	1.41 ns
NB vs ED	1.26 ns	1.53 *
CR vs UL	1.20 ns	1.09 ns
CR vs ED	1.57 *	1.90 *
UL vs ED	1.86 **	1.65 *

NB = Port Stephens; CR = Cronulla; UL = Ulladulla; ED = Eden

Results of a posteriori comparisons between sites (locations), showing *t*-values and significance levels.

Location	Sites	Fringe	Barrens
NB	1 vs 2	1.66 **	2.00 **
	1 vs 3	1.07 ns	1.38 *
	2 vs 3	1.76 *	2.30 **
CR	1 vs 2	1.32 ns	1.51 *
	1 vs 3	1.07 ns	2.30 **
	2 vs 3	1.63 **	1.78 **
UL	1 vs 2	1.34 *	1.36 ns
	1 vs 3	1.39 *	1.30 ns
	2 vs 3	1.08 ns	1.18 ns
ED	1 vs 2	1.73 **	2.16 **
	1 vs 3	0.77 ns	1.85 **
	2 vs 3	1.23 ns	2.01 **

Table 4. Taxa responsible for dissimilarity in assemblage composition between fringe and barrens habitat in each location (top five ranking taxa only shown).

Nelson Bay	Cronulla	Ulladulla	Eden
Amphipods	*Pagurus lacertosus*	*Pagurus lacertosus*	*Pagurus lacertosus*
Anabathron lene	Amphipods	Amphipods	Amphipods
Polychaetes	*Clanculus clangulus*	*Clanculus brunneus*	*Clanculus brunneus*
Pagurus lacertosus	*Clanculus brunneus*	*Rissonia fasciata*	*Rhynchocinetes serratus*
Rhynchocinetes serratus	*Agnewia tritoniformis*	*Rhynchocinetes serratus*	*Athanas granti*

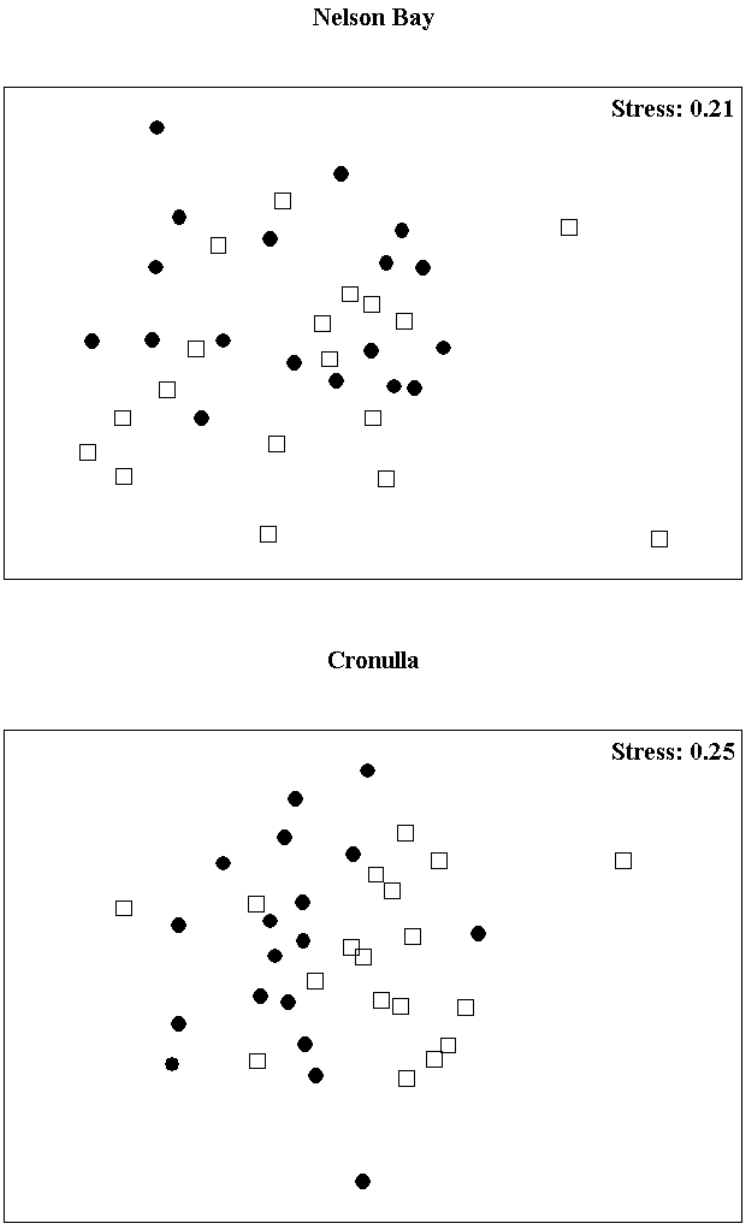

Figure 5. nMDS ordinations comparing macrofaunal assemblages in fringe and barrens habitats at each location: ● = fringe habitat; ❑ = barrens habitat. Symbols represent site averages (n=6).

Temporal changes in assemblages

Assemblages in each site varied over time (Fig. 6); however, assemblages varied in different ways in each site (Table 5). Interestingly, assemblages changed significantly over the shortest time interval (between 27 and 28 June 2001) at both sites. SIMPER analyses showed that assemblages were between 68% and 74% dissimilar between sampling times. Many taxa contributed to temporal differences between assemblages with most taxa only contributing small percentages to total dissimilarity (Table 6). Changes in abundance of the hermit crab *Pagurus lacertosus* contributed most to dissimilarity (average of 10%) while other taxa generally contributed much less than 10% each.

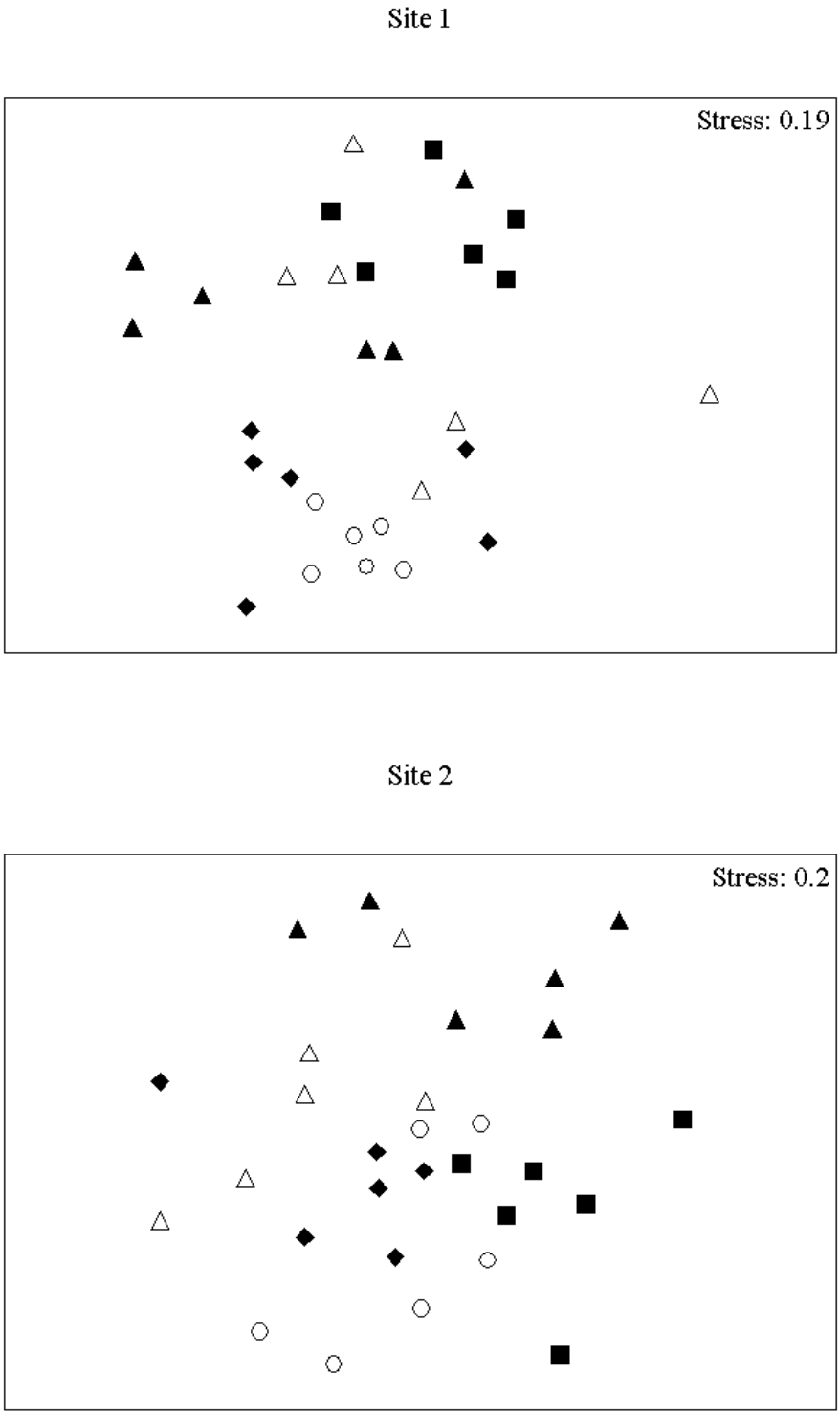

Figure 6. nMDS ordinations of temporal changes in faunal assemblages sampled from beneath *C. rodgersii* from two sites (Shelly Point, Jibbon) at Cronulla. ▲ = 20 Sept 2000; Δ = 11 Dec 2000; ■ = 27 June 2001; ◆ = 28 June 2001; ○ = 12 July 2001.

Table 6. Average dissimilarity (%) contributed by each species from comparisons between assemblages over five sampling times at Cronulla. * indicates species making highest contributions.

Taxa	Average % Contribution	Taxa	Average % Contribution
Agnewia tritoniformis	3.8	*Ischnochiton smarragdinus*	2.1
Alope australis	1.7	*Ischnochiton versicolor*	2.2
Amphithalamus incidatus	3.3	Isopod #1	1.7
Ampipods*	5.4	*Ophiactis resiliens*	0.9
Anabathron conabulatum	2.6	*Ornithochiton quercinus*	1.3
Apasmogaster costatus	1.8	Ostracod # 2	0.9
Aplysia sydneyensis	1.5	Ostracod# 1	1.3
Astralium tentoriiformis	1.9	*Pagurus lacertosus**	9.8
Athanas granti	3.4	*Pagurus sinuatus*	1.8
Cacozeliana granaria	1.7	*Patelloida mufria*	3.4
Caprellid sp.	1.1	*Pisinna sp.*	2.2
Chiton jugosus	4.2	Polychaete worms	3.2
Clanculus brunneus	3.7	*Rhynchocinetes serratus*	3.7
*Clanculus clangulus**	6.0	*Rissoina fasciata*	1.5
Clarkoma pulcra	3.6	*Tricolia sp.*	1.2
Cordita excavata	1.3	Unknown bivalve # 2	1.0
Foramnifera	4.0	Unknown gastropod # 1	2.0
Granata imbricata	0.7	Unknown gastropod # 2	2.0
Haliotis coccoradiata	2.4	Unknown gastropod # 3	1.4
Haliotis sp.	0.8	Unknown gastropod # 4	1.6
Heliocidaris eurythrogramm	1.9	Unknown gastropod # 5	2.2
Ischnochiton australis	1.2	Unknown gastropod # 6	1.5
Ischnochiton elongatus	0.8	Unknown spirobid*	5.6
		Unknown trochid	2.7

Discussion

Centrostephanus rodgersii inhabiting boreholes in fringe and barrens areas of subtidal reefs in New South Wales support diverse assemblages of benthic macrofauna. Most taxa were present in low abundances and some of these may be rare. The distribution and abundance of individual taxa and whole assemblages were shown to vary spatially and temporally. This study documented spatial and temporal patterns in abundance of macrofauna beneath *C. rodgersii* for the first time and so direct comparisons with previous studies cannot be made.

Abundance of individual taxa differed between urchins at many of the sites sampled in this study. For example, numerous individuals of the gastropod *Rissoina fasciata* and the shrimp *Rhynchocinetes serratus* could be found under one urchin and not another only 1-2 m away. Similar small-scale patchiness has been shown for mobile fauna under subtidal boulders, where widespread species had large densities under a few boulders but were absent from other boulders (Chapman and Underwood 1996). Similarly, Underwood and Verstegen, 1988) found patterns of aggregation of amphipods under intertidal limpets. Fauna associated with *C. rodgersii* may respond to features of the urchin that were too small to be detected by this study. This is evident even despite care being taken to select urchins that were the same size and which had the same sized borehole to minimise such effects.

The apparent non-random and patchy distribution of individuals of several taxa combined to make whole assemblages quite different from one urchin borehole to another, and faunal assemblages of replicate urchins within sites had a low percentage similarity shown by SIMPER analyses.

Underwood and Chapman, 1996) suggest that non-random patterns in abundance indicate the scale at which underlying causative processes are operating. It should be noted that actual tests of non-random pattern was not done in this study. Andrew and Mapstone, 1987) caution against using data collected primarily to estimate abundance for tests of randomness due to several sampling artefacts. Measures of distances between organisms would be required to confirm non-random patterns (Andrew and Mapstone 1987).

Many small-scale processes could be operating to cause the assemblages under individual urchins to vary spatially and temporally. Small-scale responses to habitat have been observed for intertidal animals (Underwood and Chapman 1992; Chapman 1994; Underwood and Chapman 1996). Small-scale processes that could be operating differentially between individual urchins include the nightly movements of the associated fauna, predation, competition within the borehole, territorial and mating behaviours.

Associated fauna are mobile at night when *C. rodgersii* leaves its borehole to graze (*Personal observation*). These movements could cause them to redistribute into available habitats when they seek shelter again at daybreak. Thus, the considerable variation in abundance between individual urchins could be a reflection of their movements during the previous night. Similar nightly movements occur for epibenthic fauna in seagrass habitats (Howard 1985). Another possible explanation is that different rates of predation may occur on the fauna inhabiting different urchins. Selective predation can modify competitive interactions and increase species diversity and densities of some species (Gilinsky 1984).

Habitat preference and subsequent occupancy are known to be influenced by the densities already present in a habitat patch (Weins 1976). When space is limiting and competition is strong, the total number of individuals that can be supported for a given area is fixed (Peake and Quinn 1993). Total numbers of individuals in this study varied between replicate urchins, which indicates that competition may not be constant over time and space in the boreholes of individual urchins. Refuges can enhance territorial behaviour (Beck 1995). For example, clingfish (*Apasmogaster costatus*) were usually found as single individuals in a borehole, suggesting territoriality. Individual hermit crabs, however, were numerous when present. Separation of individuals may indicate territoriality behaviours as well as different resource use patterns (Weins 1976). Mating behaviours also influence the distribution and abundance of individuals with some species moving between adjacent urchins in search of mates (Baeza and Theil 2000). Mating behaviours would cause non-random patterns of aggregation under urchins both spatially and temporally.

There were few differences between habitats when the abundance of individual taxa were examined. The hermit crab *Pagurus lacertosus* was the only species to vary in abundance between habitats and was always more abundant in the fringe habitat. Other taxa such as Amphipods, the brittlestar *Clarkoma pulcra* and a gastropod *Clanculus brunneus* also showed variation between habitats however differences were not the same at all sites and locations. In contrast, whole assemblages of macrofauna differed between fringe and barrens habitats at the two southernmost locations (Eden and Ulladulla). Further studies will be required to determine whether these differences are consistent through time. Currently, fishers mostly harvest *C. rodgersii* from the fringe habitat, which may be of concern since more species were found here than in the barrens. The significance of any impacts from harvesting *C. rodgersii* will in part depend on the uniqueness of the resident assemblages in different habitats and, based on the results of this study, it is possible that impacts will not be uniform along the NSW coastline.

Most variation in abundance of individual taxa occurred between sites within each location. Assemblage structure also differed between sites. The dynamic nature of small-scale processes operating within a site can result in variation between sites. Other processes that may vary at larger

scales to produce between-site differences include recruitment, predation, the patchy distribution and density of *C. rodgersii*, and wave action.

Recruitment can be patchy over scales of hundreds of meters to larger scales of km (Underwood and Chapman 1996). Sites in this study were separated by ~ 2km. Small-scale differences in recruitment are unlikely to affect mobile species as they can move between habitats (Chapman 1994; Underwood and Chapman 1996; Robinson and Tully 2000). That is, lasting effects of variable recruitment are less likely within sites for mobile species, but may cause differences between sites. As species grow in size after recruitment, the importance of refuges may change (Beck 1995; Robinson and Tully 2000). Mobile animals may migrate into *C. rodgersii* refuges at different stages of their life cycle. Comparing the habitat use of taxa associated with *C. rodgersii* throughout their life cycle would allow the relative importance of different habitats to be established (Beck 1995).

Differences in assemblage structure and abundances of individual taxa could reflect variation in predator pressure. Reef fishes show large variations in abundance from place to place and over time (Lincoln Smith and Jones 1995). Predation may be a strong force regulating the assemblages under *C. rodgersii.* Two species occurring under *C. rodgersii* (the shrimp *Athanas granti* and the brittlestar *Clarkoma pulcra*), are coloured to match the urchin, suggesting strong predation pressure (P.K. Dayton, *personal communication*). However, the demonstration of predation effects by fishes is difficult (Choat 1982), because significant mortality is not apparent for many prey populations.

Urchins in the fringe habitat appeared to be more sedentary than those in the barrens habitat. Fringe urchins at Cronulla remained in their boreholes for 3 hr after nightfall while urchins in the barrens had moved out of their boreholes to feed (*Personal observation*). Some authors (Rogers-Bennett *et al.* 1995; Schoppe and Werding 1996) have suggested that urchins occurring in shallow fringe habitats are more sedentary because of wave action and turbulence and therefore they rely on drift algae for food rather than grazing. Assemblages beneath urchins in the fringe habitat may differ from those in the barrens because of a lack of disturbance normally caused when the urchins move to feed. Juveniles of *C. rodgersii* and another sea urchin, *Heliocidaris erythrogramma,* were found beneath adult *C. rodgersii* in the fringe habitat. Sedentary adult *C. rodgersii* in the fringe may provide important shelter for *C. rodgersii* and *H. erythrogramma* juveniles. More sampling at several times and places would be needed to ascertain whether assemblage differences between fringe and barrens habitats are likely to be influenced by wave action and whether these patterns are consistent.

It is possible that differences in assemblage structure between locations may be a result of differing geographical ranges for observed taxa. For example, some taxa were found in all locations while others were much more restricted and were only found in one location. The hingebeak shrimp *Rhynchocinetes serratus* was found in both fringe and barrens habitats at all locations whereas another shrimp *Hippolyte caradina* was only found in the fringe habitat at Nelson Bay. However, in this study, the apparent ranges of taxa are more likely to reflect our choice of sampling locations rather than their true geographical ranges, which normally requires systematic and extensive sampling at more than one sampling point in time, and in a range of habitats to be elucidated.

Abundance of a species may also change over the extent of its range. At the extreme margins of species' range, the abundance of individuals may be lower (Krebs 1994). For example, *Patelloida mufria* was more abundant at Ulladulla and declined towards the most northerly location (Nelson Bay) and towards the most southerly location (Eden). Variable recruitment between locations would also affect the relative abundances of different taxa. Therefore, both changes in the suite of taxa occurring at each location and differences in abundance would cause assemblage structure to vary between locations.

Many species collected in this study were present in low numbers and/or were patchily distributed, suggesting they are either naturally rare with small population sizes or found in higher abundances elsewhere on the reef. Rarity is difficult to determine without additional ecological information. Rarity is not just related to abundance, but also to how individuals are distributed throughout their range, the sizes of breeding populations (which can be much less than actual range), dispersal abilities and the connectedness of populations, recruitment, mortality and habitat flexibility or specificity (Chapman 1999). An absence of ecological data for many of the taxa collected in this study means that their rarity status is unknown.

Species vary in their degree of habitat specificity (Stoner and Livingstone 1980; Chapman 1994; Taylor and Cole 1994; Chapman 1999). Two species collected in this study, *Ischnochiton australis* and *Patelloida mufria* are known to exploit a range of microhabitats (Chapman 1999 and personal observation); however patterns of habitat use are not known for the majority of other taxa collected in this study. It is probable that some taxa exploit a range of cryptic microhabitats such as under boulders, in crevices or vegetation as well as under *C. rodgersii*. Therefore, taxa that appear to be rare could occur in substantial numbers at the scale of the entire reef. The use of a variety of cryptic microhabitats (including *C. rodgersii*) by macrofauna means that the assemblages under *C. rodgersii* could be very dynamic spatially and temporally. The majority of taxa in this study each made small contributions to the differences in assemblage structure between sites, locations, habitats and over short time periods suggesting that the assemblage at any one time or place appears to be less predictable than you would expect if these animals had strong associations with the urchin. That is, associations with the urchins are probably random or haphazard for most, with shifts occurring between many suitable cryptic habitats on the reef.

Identifying the impacts of a *C. rodgersii* fishery in terms of its effects on associated fauna in boreholes may be difficult given that more information is needed to determine the status of many species found in this study. The replication required to study species with low abundances through time may prohibit studies of rarity in the marine environment (Chapman 1999). Documentation of spatial patterns at several times for species that have low abundances and occur patchily in time and space requires a high degree of replication and sampling effort. Even here where 144 replicates were sampled at one time, less than 10 individuals were often found for most taxa. Therefore, detecting changes in the abundance of these taxa due to fishing effects would be difficult to separate from the background of natural patchiness in abundance without more extensive sampling. Further, few taxa would be useful as indicators of change in whole assemblages since assemblages were highly variable and not easily categorised by a few dominant taxa.

The presence of *C. rodgersii* adds to the habitat heterogeneity of subtidal reefs and provides shelter for many associated taxa. Removal of *C. rodgersii* via fishing could subject macrofauna in boreholes to increased predation. Fletcher, 1987) found that removal of *C. rodgersii* affected animal and algal species via altered growth, rates of recruitment and survival. The impact of urchin fisheries on associated species will in part depend on whether these species are widespread and occur in other habitats such as crevices and around the base of boulders, or whether they are restricted to *C. rodgersii* with small populations. Differences in patterns of spatial variation at different scales observed in this study will make any generalised predictions about the impacts of the fishery difficult.

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APPENDIX 6:

The role of the sea urchin *Centrostephanus rodgersii* (Agassiz) as a biogenic habitat for macrofaunal assemblages on the rocky reefs of New South Wales, Australia.

(to be submitted to *J. Exp. Mar. Biol. Ecol.)*

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Introduction

Diverse assemblages of fauna occur beneath the sea urchin *Centrostephanus rodgersii* living in boreholes, and these vary both spatially and temporally (Davis *et al.*, 2003). The same assemblages do not occur on open areas of the reef substratum. Several models to explain why fauna congregate beneath *C. rodgersii* were developed and include: 1) that the spines of *C. rodgersii* act as a complex habitat and provide shelter from predators; 2) that *C. rodgersii* provides a food resource to associated fauna.

Food and shelter resources are implicated as the main reasons that fauna associate with certain habitats elsewhere in the marine environment (Heck & Westone, 1977; Stoner & Livingstone, 1980; Heck & Orth, 1980; Coull & Wells, 1983; Connolly, 1997). However, discerning the food resource and habitat requirements for individual species is difficult because often the two resources are linked. For example, in studies of epifauna occurring on various types of marine vegetation, it has been found that the habitat usually doubles as a food resource and a refuge (Stoner & Livingstone, 1980; Coull & Wells, 1983; Taylor & Cole, 1994; Shaffer *et al.*, 1995; Hull, 1997). Further, the habitat choices of different species are likely to be caused by different food and habitat preferences, and no two species respond in the same way towards habitats. The fauna inhabiting the boreholes of *C. rodgersii* come from a range of functional groups and therefore are likely to differ in their diet and feeding strategies. Consequently, the factors that lead macrofauna to associate with *C. rodgersii* are probably species specific.

Studies of the fauna associated with some sea urchins have been made in the Mediterranean, New Zealand, the Red Sea, Caribbean Sea and Chile (Lützen, 1976; Pomeranz & Tsurnamal, 1976; Warén, 1981; Gherardi, 1991; Schoppe & Werding, 1996; Hofrichter & Patzner, 2000; Baeza & Thiel, 2000). Gherardi, 1991) completed a series of gut analyses and habitat choice experiments in the laboratory and found that the associated fauna relied on either food resources or shelter provided by the urchin *Echinometra mathaei*. Gherardi, 1991) suggests that some associated fauna may feed on bits of drift algae on which the urchin feeds. Others suggest that fauna may feed on

bits of microalgae that are released into the water column as the urchin grinds its teeth and spines on the substrate (Baeza & Thiel, 2000).

The mechanism that leads to associations being formed with sea urchins can operate at different life stages of the organism. For example, larvae may first select urchin habitats when they settle from the plankton. It is known that larvae are able to use various cues in order to settle on preferred substrates (Underwood & Chapman, 1995). Larvae of the crab associate (*Liopetrolisthes mitra*) are not found in alternative habitats, and as such these larvae probably prefer sea urchins as a habitat (Baeza & Thiel, 2000). For other species, the selection of sea urchins as a habitat may occur later in life when they outgrow other habitats. Many crustaceans are very mobile and show such ontogenetic shifts from less favourable habitats (Robinson & Tully, 2000). The selection of urchin habitat may also occur after the organism returns from foraging activities at night. When the fauna return to shelters (daybreak), several cues (chemical or visual) may be used to locate urchin habitats (Gherardi, 1991).

This experiment aims to test the hypotheses that: 1 The abundance of selected macrofauna, species richness, species diversity, total individuals and structure of assemblages will be greater under *C. rodgersii* and mimic urchins compared to rocks and equal between *C. rodgersii* and mimic urchins, and, 2. The abundance of selected taxa, species richness, species diversity, total individuals structure of assemblages will be greater under *C. rodgersii* than compared to mimic urchins and rocks and will be equal between mimic urchins and rocks.

Methods

Experimental design and sampling

This experiment was conducted in the fringe habitat at two sites (Shelly Point and Jibbon) and two times (28/06/01 and 12/07/01) at Cronulla, NSW. There were 4 treatments with 6 replicates of each treatment in each site and time (a total of 96 replicates). Descriptions of the treatments are as follows:

Mimic Urchin. A mimic urchin was constructed from plastic balls and bamboo skewers and coloured to resemble a real *C. rodgersii* urchin. The treatment was designed to mimic the refuge characteristics (spine canopy) of a real urchin but without providing any food resources to the associated fauna (Figure 1).

Natural Urchin. Sampling macrofauna from beneath *Centrostephanus rodgersii* to compare these assemblages with other treatments.

Control (Disturbed Urchin). Removal of a real urchin from its borehole and replacing it again. This treatment served to act as a control procedure for treatment 1. The control estimates the effect of disturbance that would occur to associated fauna when replacing real urchins with mimic urchins.

Rocks. Sampling macrofauna from the underside of rocks. The underside of rocks represents a habitat that neither offers a spine canopy or food resources.

Figure 1. Photograph of a mimic urchin positioned within the borehole of a *C. rodgersii* urchin. Photo shows metal arm (top) extending as the attachment point to the substrate.

Mimic urchin and disturbance control treatments were set-up within one day on the 27/06/01. Natural urchin and rock treatments required no preparation and were simply sampled on the subsequent sampling days. After treatments were set up, a map indicating the position of mimic urchins and disturbed urchins were drawn on a slate so that they could be found again for subsequent sampling. Treatments were interspersed randomly within the site (Hurlbert, 1984).

Mimic urchins were fastened securely to the substrate by a metal arm projecting from the base of the urchin. An underwater ramset gun was used to bolt the metal arm of the mimic urchin onto the substrate adjacent to the borehole of a real urchin. The real urchin was carefully removed from the borehole and the mimic urchin was positioned inside the borehole to cover the associated fauna. For the disturbance control, *C. rodgersii* were selected and removed for a brief period and then replaced in their borehole again. Urchins were held in position until they had reattached within the borehole.

Sampling was carried out at 2 times after the initial set-up of the treatments (at 24hrs and 15 days). However, because of large seas following the first sampling day, 3 mimic urchin replicates did not remain at one site upon inspection on the second sampling day, and were not collected. Macrofauna were collected by divers on SCUBA using a suction sampling device known as an "air lift" with a 180 μm mesh bag attached. Urchins were selected haphazardly in the fringe habitat from depths of around 4 m. Fringe habitat was determined by depth and the presence of geniculate coralline algae and the foliose algae *Ecklonia radiata and Sargassum sp*. Fauna were collected from below solitary urchins, that is, those whose shelter was a single eroded hollow in the rock substrate. Urchins of ~ 100 mm test diameter were sampled. By selecting solitary urchins of a similar size, the area of substrate sampled was consistent among replicates since the size of the urchin dictates the size of the borehole.

Samples were washed through a 0.5 mm sieve, sorted, counted and identified wherever possible to species. Specimens unable to be identified were classified as different 'morphospecies' (Oliver and Beattie 1996) within classes.

Univariate statistical analyses

Because only three mimic urchins (instead of 6 replicates) were sampled from one site at the second sampling time, data for a fourth replicate was generated by taking an average of the other

three (Underwood, 1997). Generating a fourth replicate allowed for balanced analyses (using 4 replicates for each treatment at each site and time).

The removal of sea urchins via fishing may impact on any species that rely on the urchin for habitat or food resources. Therefore, the choice of taxa to include in univariate analyses (ANOVA) was made according to which ones were present consistently under *C. rodgersii* in each of the Malacostraca, Gastropoda, Polychaeta and Polyplacophora classes. Three additional taxa were analysed (the shrimp *Athanas granti*, the clingfish *Apasmogaster costatus* and the brittlestar *Clarkoma pulchra)* because they were identified from the literature as potential species to form associations with *C. rodgersii*. Additionally, species richness, total numbers of individuals and species diversity were analysed. Significant differences were further investigated using Student Neuman-Keuls (SNK) tests to identify differences between pairs of means.

The factor of Treatment was fixed, Time was fixed and orthogonal, and Sites were random and orthogonal. Time was treated as a fixed factor because the study was conducted over a very short time interval and thus would not represent a random sample from all possible times (see Kingsford, 1998).

Data were tested for homogenous variances using Cochran's Test prior to analysis and wherever necessary data were transformed using the Log (x +1) transformation to homogenise variances (Underwood, 1981). In some instances, transformations of the data were unsuccessful and analyses were done using data with heterogeneous variances because ANOVA is robust to departures from homogeneity (Underwood, 1981). Analysing data with heterogeneous variances increases the chance of Type I error, therefore, for these analyses a probability level of $\alpha = 0.01$ was used (Underwood, 1981).

Multivariate statistical analyses

Multivariate analyses were done to determine whether the assemblage of macrofauna differed between treatments using PRIMER v5 statistical package (Clarke & Warwick, 1994). All available replicates were used for multivariate analysis because there is no restriction to have equal numbers of replicates (Clarke & Warwick, 1994). Further, increased replication increases the number of possible permutations and reduces the Type 1 error associated with making several pair-wise comparisons (Clarke & Warwick, 1994). The mimic urchin treatment used 6 replicates at the first sampling time and 4 replicates at the second sampling time. For all other treatments, 6 replicates were used at each time and site. Data were fourth-root transformed before creating similarity matrices (Bray-Curtis) because some taxa were very abundant. Analysis of similarities (ANOSIM), species contributions to similarity/dissimilarity (SIMPER) and non-metric multidimensional scaling (nMDS) were used to show any patterns in the faunal assemblages from various treatments. The average percentage contributions that each species made to dissimilarity were calculated to determine which species were most important in distinguishing between treatments.

Results

General findings

A total of 7513 individuals representing 123 taxa were sampled in this study (Table 1). Most individuals came from the classes Gastropoda and Malacostraca but the dominance of these classes varied between treatments (Fig 2). The percentage of individuals representing the different classes was similar between natural urchins and disturbed urchins (control). Mimic urchins showed an increase in the percentage of gastropods and a decrease in Malacostraca. The percentage of each class from under rocks appeared to be more similar to mimic urchins than to natural urchins.

Of the 36 taxa with less than 10 individuals found, 18 were in higher numbers under rocks, 5 had lower abundance under rocks and 13 were not found under rocks. The clingfish (*Apasmogaster costatus*) was the only species with low abundance that did not occur under rocks, it's habitat specificity is unknown, and it could rely on the urchin for habitat. Other taxa thought likely to form an association with the urchin were *Clarkoma pulchra* and *Athanas granti*. *Athanas granti* was found under rocks confirming that this species does exist in another habitat. However, the brittlestar was absent from beneath rocks.

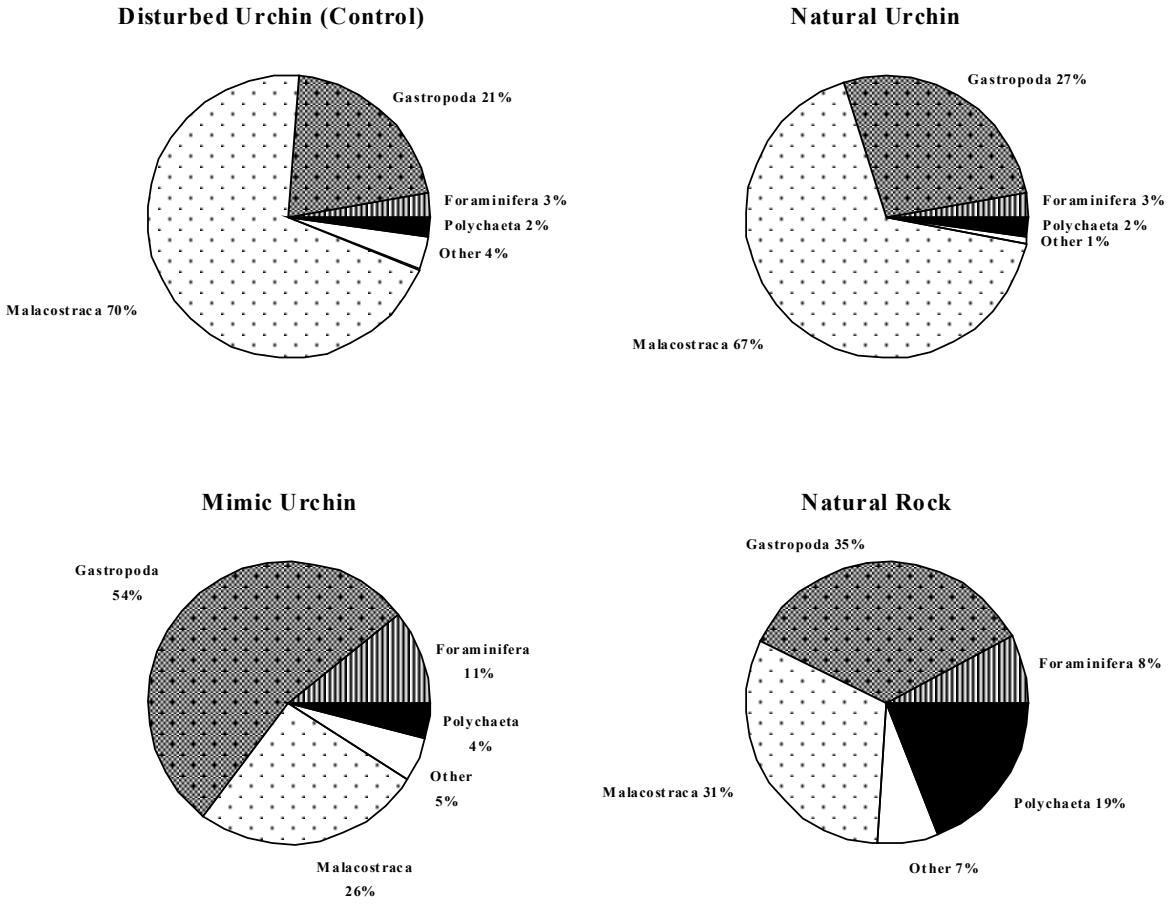

Figure 2. The percentage of taxa representing different classes in each treatment.

Table 1. Summary of analysis of variance results for species diversity, species richness, total individuals and the abundance of selected taxa between treatments at Cronulla, NSW in July 2001. (n=4). A [1] indicates data are LN(x +1) transformed; C = Cochran's test; ns = not significant (P >0.05); * = significant (P <0.05) and ** significant (P <0.01).

		Species Diversity C = 0.1776ns		Species Richness C = 0.8678**		Total Individuals C = 0.3520**		Polychaete Worms[1] C = 0.1973ns	
Source	**df**	**MS**	**F**	**MS**	**F**	**MS**	**F**	**MS**	**F**
Treatment (Tr)	3	3.34	7.58	424.68	2.93	16388.52	2.62	14.75	10.74*
Time (Ti)	1	0.11	0.08	328.52	0.36	21425.64	0.96	0.06	0.03
Site (Si)	1	0.28	1.92	2537.64	15.54**	316546.89	46.78**	8.48	15.26**
Tr x Ti	3	0.07	0.14	326.43	0.82	9122.68	0.91	0.80	0.69
Tr x Si	3	0.44	2.99*	144.89	0.89	6253.68	0.92	1.37	2.47
Ti x Si	1	1.34	9.11**	922.64	5.65	22238.27	3.29	1.75	3.15
Tr x Si x Ti	3	0.50	3.38*	397.39	2.43	10001.72	1.48	1.16	2.08
Residuals	48	0.15		163.35		6766.67		0.56	
Total	63								
		Amphipoda[1] C = 0.2747ns		*Rhynchocinetes serratus*[1] C = 0.2613ns		*Pagurus lacertosus*[1] C = 0.4848ns		*Ischnochiton versicolor*[1] C = 0.2561ns	
Source	**df**	**MS**	**F**	**MS**	**F**	**MS**	**F**	**MS**	**F**
Treatment (Tr)	3	8.92	41.98**	0.09	0.47	11.63	3.78	0.22	1.31
Time (Ti)	1	0.01	0.02	0.05	0.32	0.05	0.03	1.06	1.06
Site (Si)	1	21.68	36.62**	0.16	0.82	73.15	65.77**	3.42	10.33**
Tr x Ti	3	2.97	2.19	0.99	19.33*	1.93	1.73	0.17	7.91
Tr x Si	3	0.21	0.36	0.20	1.05	3.07	2.76	0.17	0.51
Ti x Si	1	0.36	0.60	0.16	0.82	1.86	1.67	1.0	3.01
Tr x Si x Ti	3	1.35	2.29	0.05	0.27	1.11	1.00	0.02	0.06
Residuals	48	0.59		0.19		1.11		0.33	
Total	63								
		Ischnochiton australis[1] C = 0.2692ns		*Clanculus clagulus*[1] C = 0.1316ns		*Agnewia tritoniformis*[1] C = 0.1662ns		*Clarkoma pulchra*[1] C = 0.2603ns	
Source	**df**	**MS**	**F**	**MS**	**F**	**MS**	**F**	**MS**	**F**
Treatment (Tr)	3	0.31	1.05	2.04	0.79	0.21	0.51	1.53	1.23
Time (Ti)	1	0.23	0.54	8.38	4.74	0.57	1.35	0.29	2.19
Site (Si)	1	2.32	22.71**	0.04	0.06	0.03	0.07	6.21	36.31**
Tr x Ti	3	0.04	2.99	0.29	2.86	0.62	4.30	0.15	2.78
Tr x Si	3	0.29	2.88*	2.59	3.72*	0.41	1.02	1.25	7.29**
Ti x Si	1	0.43	4.22*	1.77	2.53	0.42	1.04	0.13	0.76
Tr x Si x Ti	3	0.01	0.13	0.10	0.14	0.14	0.35	0.05	0.31
Residuals	48	0.10		0.70		0.40		0.17	
Total	63								
		Athanas granti[1] C = 0.2336ns		*Apasmogaster costatus* C = 0.2416ns					
Source	**df**	**MS**	**F**	**MS**	**F**				
Treatment (Tr)	3	0.09	0.67	0.72	1.53				
Time (Ti)	1	0.90	17.86	0.04	1.0				
Site (Si)	1	0.55	3.68	0.09	0.33				
Tr x Ti	3	0.13	0.22	0.10	0.78				
Tr x Si	3	0.13	0.86	0.47	1.83				
Ti x Si	1	0.05	0.33	0.04	0.17				
Tr x Si x Ti	3	0.58	3.82*	0.13	0.49				
Residuals	48	0.15		0.26					
Total	63								

Univariate statistical analyses

For the two hypotheses proposed by this experiment to be supported, treatments needed to differ to each other in a set pattern. However differences between treatments only occurred at certain times or sites, shown by significant lower order interaction terms. For these species, treatment means were compared using SNK tests to identify if whether shelter requirements or food resources (Hypotheses 1 & 2) could be implicated for the higher abundance under *C. rodgersii*. The abundance of *Clarkoma pulchra* did vary between treatments in such a way that the food resource hypothesis could be supported (Table 2). For instance, the abundance of the brittlestar was higher under real urchins than either the rock or mimic treatments, both of which do not provide food resources. Also, the mimic and rock treatments were similarly low in the abundance of this brittlestar. These patterns of difference between treatments allowed the null hypothesis to be rejected and the hypothesis relating to food resources (hypothesis 2) accepted. However, it must be cautioned that since this result was only demonstrated at site 1, there can be no generality inferred from this result and the model only holds for one site at Cronulla. Another species (the hermit crab *Pagurus lacertosus*) came very close to being significantly different in abundance between treatments in site one, which would have also supported the second hypothesis (treatment x site $P = 0.0526$). For all other taxa analysed, no clear differences between treatments that could support either hypothesis were identified.

Table 2. Results of ANOSIM and SIMPER analyses for treatment comparisons in each time and site. Global R = test statistic for overall differences between treatments at a single time and site combination. R-stat indicates significance of pairwise comparisons. %D = the percentage dissimilarity between treatment pairs. An * shows significant effects (P <0.05) and ** indicates significance at (P <0.01).

	Global R	Mimic & Urchin		Mimic & Rock		Urchin & Control		Urchin & Rock	
		R-stat	%D	R-stat	%D	R-stat	%D	R-stat	%D
Time 1 Site 1	0.38**	0.111ns	65	0.517**	74	0.235ns	51	0.781**	66
Time 1 Site 2	0.534**	0.753*	54	0.519*	58	0.331**	51	0.665**	52
Time 2 Site 1	0.282**	0.246*	69	0.465*	74	0.125ns	70	0.329*	72
Time 2 Site 2	0.24**	0.496**	71	0.095ns	59	0.166ns	68	0.269*	72

Species diversity (H) showed a significant interaction between treatments, times and sites (Table 2, Figure 3). Species richness and total individuals were significantly different between sites but not treatments or times (Table 2, Figure 3).

The abundance of polychaete worms and amphipods were significantly different between treatments and sites (Table 2). Both polychaete worms and amphipods were significantly higher under rocks than any other treatment and both were higher in site 1 than at site 2 (Figure 3).

The abundance of the hingebeak shrimp (*Rhynchocinetes serratus*) interacted significantly between treatment and time (Table 2). At the first sampling time their abundance under natural urchins was higher than any other treatment, this was because they were absent from under rocks at this time. At the second sampling time there was no significant difference between treatments, although they were absent from mimic and real urchins (Figure 3).

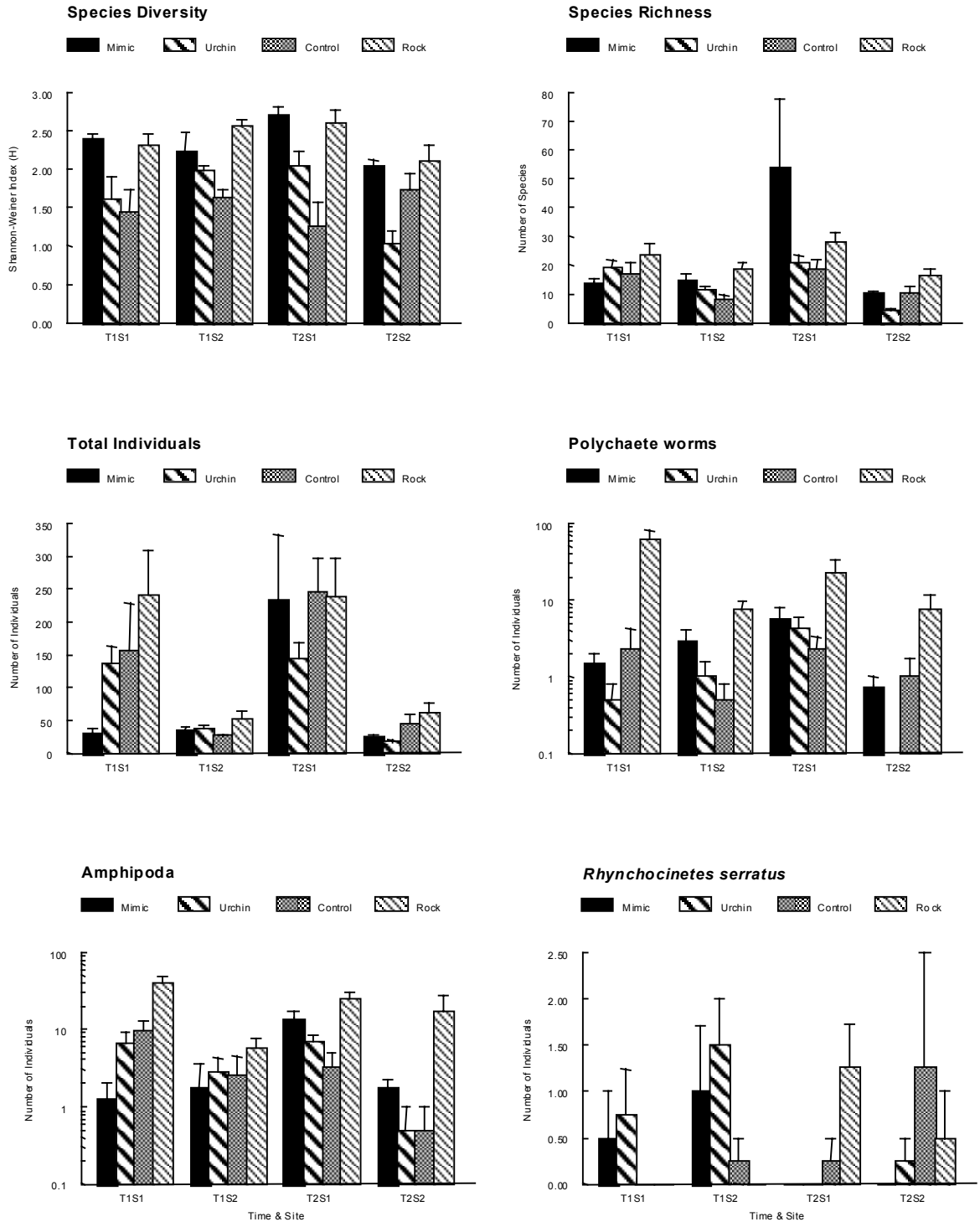

Figure 3. Average species diversity, species richness, total individuals and abundance of 11 selected taxa for each treatment in each of 2 sites (site 1 = Shelly Point; Site 2 = Jibbon) and times (28/06/01 and 12/07/01) at Cronulla in July, 2001 (± SE, n = 4). Note logarithmic scale used for polychaete worms, amphipoda and *Pagurus lacertosus*.

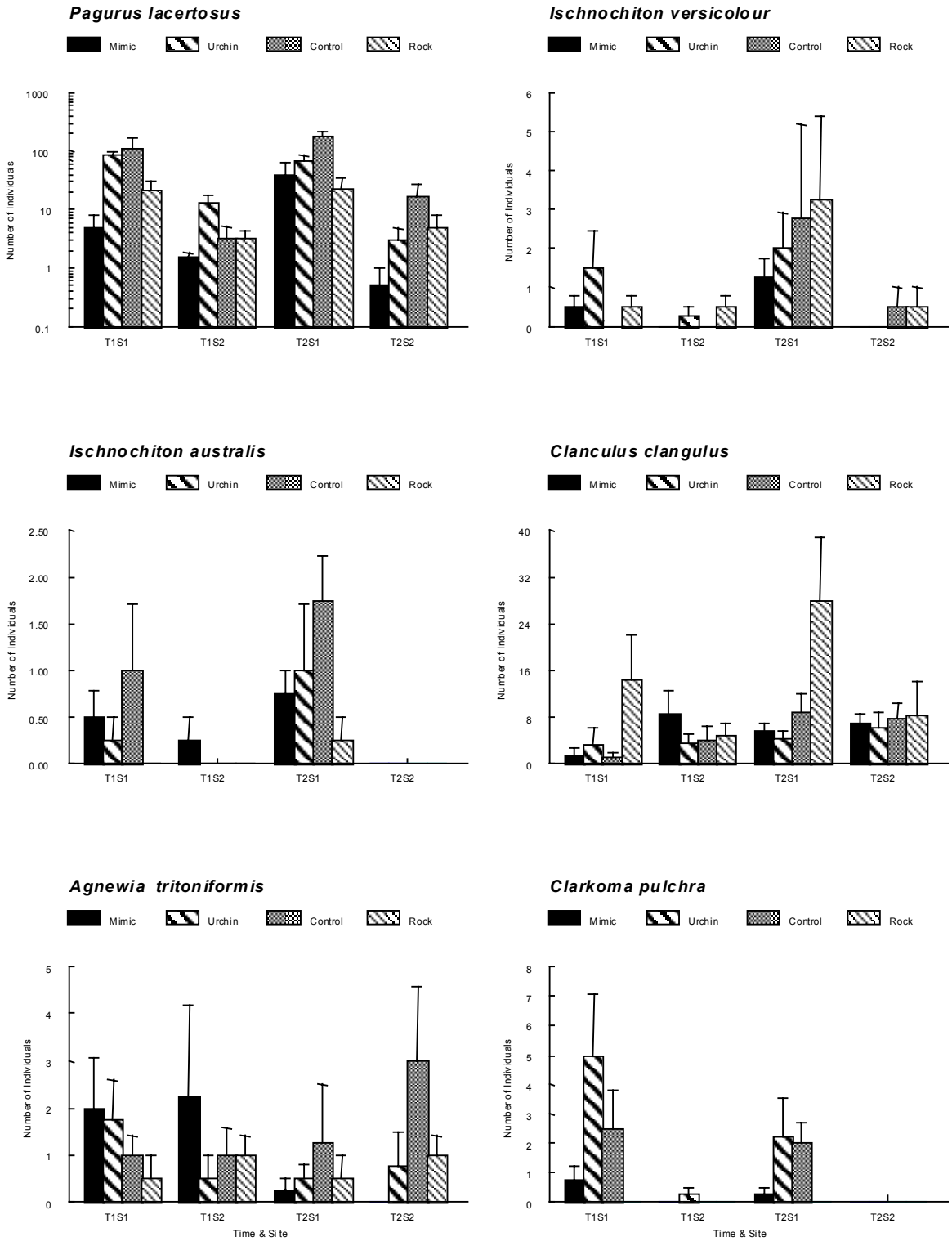

Figure 3. (Continued) Average species diversity, species richness, total individuals and abundance of 11 selected taxa for each treatment in each of 2 sites (site 1 = Shelly Point; Site 2 = Jibbon) and times (28/06/01 and 12/07/01) at Cronulla in July, 2001 (± SE, n = 4). Note logarithmic scale used for polychaete worms, amphipoda and *Pagurus lacertosus*.

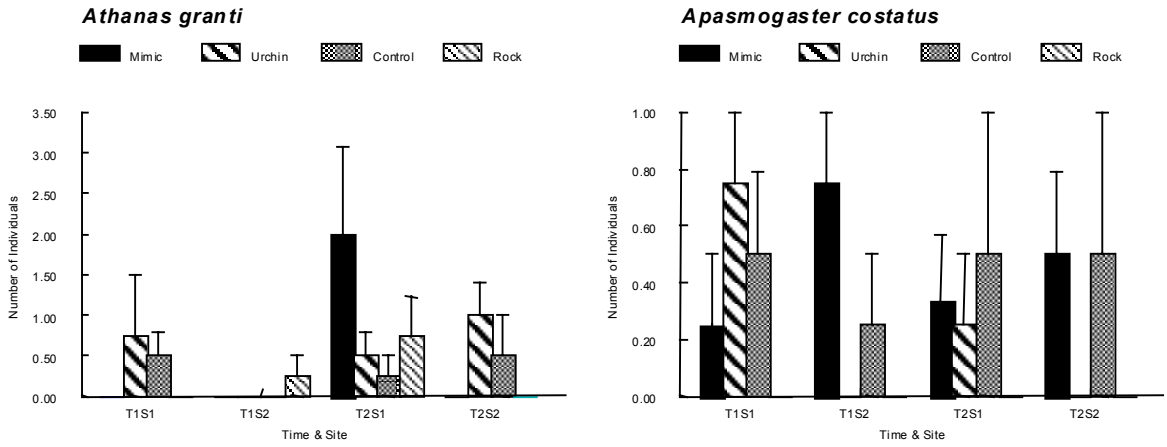

Figure 3. (Continued) Average species diversity, species richness, total individuals and abundance of 11 selected taxa for each treatment in each of 2 sites (site 1 = Shelly Point; Site 2 = Jibbon) and times (28/06/01 and 12/07/01) at Cronulla in July, 2001 ($\pm$ SE, n = 4). Note logarithmic scale used for polychaete worms, amphipoda and *Pagurus lacertosus*.

Both the hermit crab (*Pagurus lacertosus*) and the chiton *(Ishnochiton versicolor*) varied in abundance between sites but not between treatments or time (Table 2). Both species were higher in abundance at site 1 (Figure 3). Another chiton (*Ishnochiton australis*) showed significant interactions between treatment with site and time with site (Table 2). At site 1, the control treatment was significantly higher than all other treatments. This pattern also occurred at site 2 but results were not significant. The abundance of this species also varied temporally with no significant differences between sites at the first sampling time but with site 2 being significantly higher at the second sampling time (Figure 3).

The gastropod *Clanculus clangulus* and the brittlestar *Clarkoma pulchra*, showed significant interactions between treatment and site (Table 2). The gastropod was significantly higher under rocks than any other treatment at site 1, but there was no difference at site 2 (Figure 3). The brittlestar showed no significant difference in abundance between treatments at site 2, however there was differences at site 1. The brittlestar was more abundant under real urchins than compared to rocks (where it was absent) and mimic urchins. Mimic urchins and rocks were not significantly different to each other. The control treatment and the real urchin treatment were also not significantly different to each other. This combination of differences between treatments for *Clarkoma pulchra* supports hypothesis 2, therefore the null was rejected, and the food resource model (model 2) supported.

The shrimp *Athanas granti* showed a significant treatment, site and time interaction however SNK tests failed to indicate specific differences between treatments at any particular time or site (Table 1). This species was found under *C. rodgersii* as well as under rocks but the abundances were so low that averages were less than one individual (Figure 3).

Two species, the whelk *Agnewia tritoniformis* and the clingfish *Apasmogaster costatus* showed no significant differences between any of the factors of treatment, time or site (Table 2). This whelk had large standard errors associated with mean abundance (Figure 3). The clingfish was actually absent from some treatments and times, and even when it was present, the abundance averaged less than 1 individual per replicate (Figure 3).

Multivariate statistical analyses

Comparison between treatments

Non-metric multidimensional scaling ordination (nMDS) was done on the abundances of all taxa from each treatment and at each combination of site and time. Differences between the assemblages of treatments were most clearly represented by the nMDS plots of both sampling times at Shelly Point (Fig 4 a-b). The rock treatment and replicates at Shelly Point were grouped together (Fig 4 a-b). There appeared to be no differences between the natural urchin and control treatments because the replicate points are overlapping (Fig 4 a-b). Generally, treatments at Jibbon were more similar than at Shelly Point (Fig 4 a-d). Mimic treatment replicates were also similar to each other at the second sampling time at Jibbon but not at Shelly Point in the first sampling time (Fig 4 a-d).

ANOSIM tests confirmed that the assemblages of the treatments were significantly different to each other overall for each site and time combination (Table 3). Pairwise comparisons of the assemblages between mimic and natural urchin treatments showed significant differences except at the first sampling time at Shelly Point (Table 3). Subsequent SIMPER analyses showed that these treatments ranged from being 54% to 71% dissimilar. Pairwise comparisons of mimic and rock treatments showed significant differences of assemblages except at the second sampling time at Jibbon (Table 3). Mimic urchins and rocks were between 58% and 74% dissimilar (Table 3). Natural urchins and the disturbance control urchins were not significantly different except at the first sampling time at Jibbon (Table 3). The average dissimilarity between these treatments ranged between 51% and 70%. The assemblages of rock and natural urchin treatments were significantly different to each other at all time and site combinations and ranged from being 52% to 72% dissimilar (Table 3).

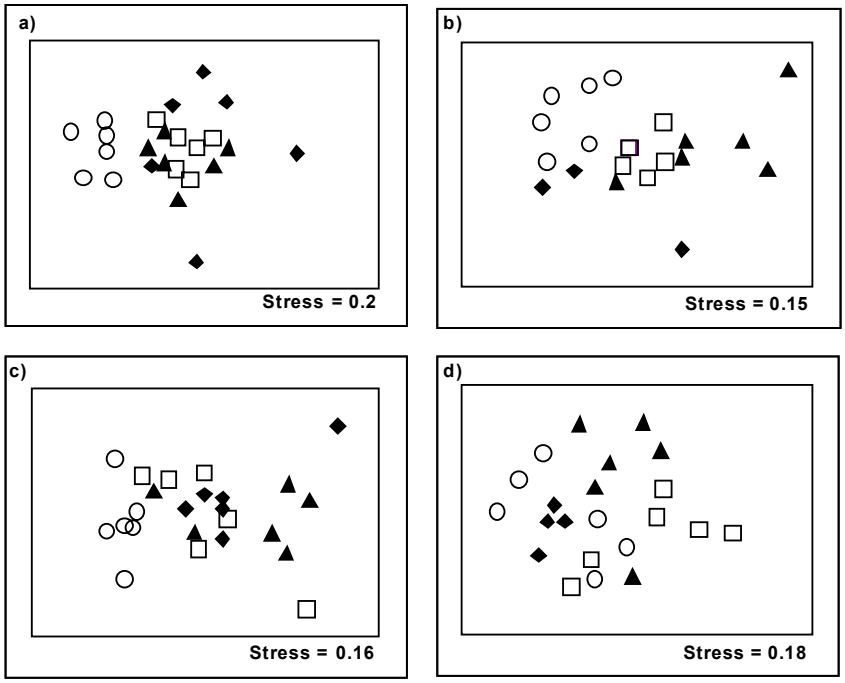

Figure 4. Non-metric multidimensional scaling (nMDS) ordination of faunal assemblages sampled from 4 treatments. Abundances are fourth root transformed. Comparison between treatments a) 28/06/01 Shelly Point, b) 12/07/01 Shelly Point, c) 28/06/01 Jibbon, d) 12/07/01 Jibbon, ◆ = Mimic; □ = Natural Urchin; ▲ = Control; = Rock.

Table 3. Results of ANOSIM and SIMPER analyses for comparisons through time and between sites for each treatment. R-stat indicates significance of pairwise comparisons. %D = the percentage dissimilarity between treatment pairs. An * shows significant effects (P <0.05) and ** indicates significance at (P <0.01).

	Through Time				Between Sites			
	Site 1		Site 2		Time 1		Time 2	
	R-stat	%D	R-stat	%D	R-stat	%D	R-stat	%D
Mimic	0.191ns	73	0.226ns	59	0.306**	75	0.926*	71
Urchin	0.326**	51	0.268*	72	0.501**	71	0.774**	79
Control	0.363**	58	0.013ns	67	0.674**	79	0.476**	69
Rock	0.152ns	52	0.122ns	67	0.106ns	59	0.25*	62

At time 1 Site 1, assemblages from under rocks were different to both real urchins and mimic urchins and the mimic urchin and real urchin were not significantly different (Table 3). This result supports hypothesis 1 (shelter model) since the rock does not provide a spiny canopy like the mimic urchin and the real urchin. At time 2 site 2, hypothesis 2 (food model) was supported because real urchins were significantly different to rocks and mimics but rocks and mimic were not different to each other (Table 3).

The top five species contributing to the observed dissimilarity between treatments varied for each pair-wise comparison and at different times and sites (Table 4). The remainder of the species (generally more than 30 additional species) each contributed less than 3% towards dissimilarity.

Table 4. List of taxa collected from each of 4 experimental treatments at Cronulla in July, 2001.

Taxa	Urchin mimic	Natural urchin	Disturbed Urchin (Control)	Rock
Gastropoda				
Agnewia tritoniformis	X	X	X	X
Amphithalamus incidatus	X	X	X	X
Anabathron conabulatum	X	X	X	X
Anabathron lene	X			X
Anabathron luteofuscus	X	X		X
Aplysia sydneyensis	X	X		
Astralium tentoriiformis	X	X	X	X
Bullina lineata	X			
Cabestana spengleri		X		X
Cacozeliana granaria	X	X	X	X
Charonia lampas rubicunda			X	
Clanculus brunneus	X	X	X	X
Clanculus clangulus	X	X	X	X
Cominella eburnea	X			
Conus aplustre	X	X		
Cyprea sp			X	
Diodora lineata	X		X	X
Eulumnid sp		X		X
Gena impertusa	X	X		X
Granata imbricata	X	X	X	
Haliotis coccoradiata	X	X	X	X
Herpetopoma aspersa	X	X	X	X
Littorina unifasciata				X
Mitra badia				X
Mitrella semiconvexa		X		X
Mitrella tayloriana	X			X
Nassarius pauperrus	X			
Notoacmea petterdi	X			
Patelloida altocostata		X		
Patelloida mufria	X	X	X	X
Phasianotrochus eximius	X			
Pisinna sp.	X	X	X	X
Ranella australis		X		
Rissoina fasciata	X	X	X	X
Terebellid sp	X			X
Terebra jacksonian		X		
Tricolia sp.	X	X	X	X
Unknown gastropod # 1	X	X	X	
Unknown gastropod # 2	X	X	X	X
Unknown gastropod # 3			X	
Unknown gastropod # 4	X	X	X	X
Unknown gastropod # 5				X
Unknown gastropod # 6		X	X	X
Unknown gastropod # 7	X	X	X	X
Unknown snail # 1	X	X	X	X
Unknown Snail # 2	X	X	X	X
Unknown Spirobid		X		

Table 4. (Continued) List of taxa collected from each of 4 experimental treatments at Cronulla in July, 2001.

Taxa	Urchin mimic	Natural urchin	Disturbed Urchin (Control)	Rock
Polyplacophora				
Chiton jugosus	X	X	X	
Cryptoplax mystica		X		
Cryptoplax striata				X
Ischnochiton australis	X	X	X	X
Ischnochiton elongatus		X	X	X
Ischnochiton smarragdinus	X	X	X	X
Ischnochiton versicolor	X	X	X	X
Ornithochiton quercinus	X	X	X	
Unknown Chiton # 1				X
Unknown Chiton # 2			X	
Bivalvia				
Barbatia riculata				X
Cordita excavata	X	X		X
Limaria orientalis				X
Marikellia solida			X	
Pinctada sp	X			
Scaeochlamys lividus				X
Unknown Bivalve # 1	X		X	X
Unknown Bivalve # 2	X		X	X
Anthozoa				
Actinia tenebrosa				X
Aulactinia veratra			X	
Asteroidea				
Allostichaster polyplax	X			X
Pateriella sp		X	X	
Ophiuroidea				
Brittlestar # 1				X
Brittlestar # 2				X
Brittlestar # 3				X
Clarkoma canaliculata			X	
Clarkoma pulcra	X	X	X	
Ophiactis resiliens	X	X	X	X
Ophiarachnella ramsayi				X
Ophionereis schayerii	X			X
Ophiothrix caespitosa			X	X
Crinoidea				
Crinoid unknown sp	X			
Echinoidea				
Centrostephanus rodgersii	X	X	X	
Heliocidaris erythrogramma	X	X	X	X
Holopneustes pycnotilus			X	

Table 4. (Continued) List of taxa collected from each of 4 experimental treatments at Cronulla in July, 2001.

Taxa	Urchin mimic	Natural urchin	Disturbed Urchin (Control)	Rock
Malacostraca				
Actaea peronii	X			
Alope australis		X	X	
Amphipods	X	X	X	X
Athanas . sp		X	X	X
Athanas granti	X	X	X	X
Caprellid. sp		X		X
Crab # 1	X		X	X
Crab # 2	X			X
Crab # 3	X	X		X
Crab # 4	X			
Crab # 5	X			X
Cumacean		X		X
Isopod #1		X	X	X
Isopod #2		X		
Mysiid. Sp		X	X	X
Nebalia. sp	X			
Pagurus lacertosus	X	X	X	X
Pagurus sinuatus	X	X	X	X
Pilumnus tomentosus				X
Plagusia chabrus	X			X
Rhynchocinetes serratus	X	X	X	X
Unknown shrimp # 1				X
Ostracoda				
Ostracod # 1	X	X	X	X
Ostracod # 2	X	X	X	X
Cirripedia				
Tesseropora rosea				X
Chelicerata				
Unknown mite		X	X	
Unknown sea spider				X
Nemertean				
Nemertean (unknown. sp)				X
Platyhelminthes				
Flatworm				X
Sipuncula				
Phascolosoma noduliferum				X
Polychaeta				
Diapatra dentata				X
Eunicidae sp.				X
Lepidonotus melanogrammus	X			

Table 4. (Continued) List of taxa collected from each of 4 experimental treatments at Cronulla in July, 2001.

Taxa	Urchin mimic	Natural urchin	Disturbed Urchin (Control)	Rock
Osteichthyes				
Apasmogaster costatus	X	X	X	
Enneapterygius rufopileus		X		
Heteroclinis eckloniae	X			
Foraminifera	X	X	X	X

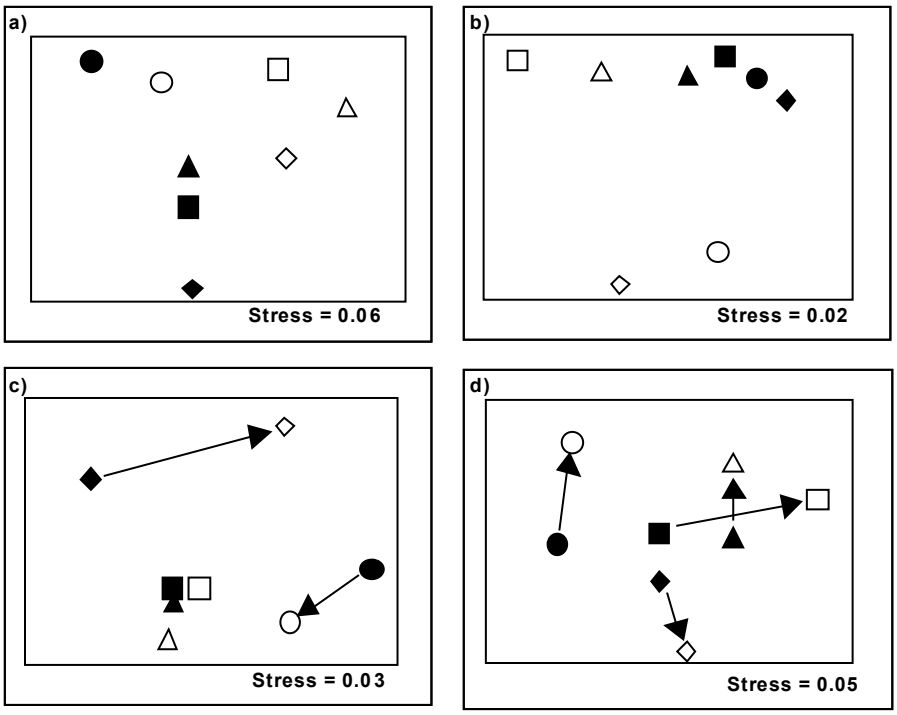

Figure 5. Non-metric multidimensional scaling (nMDS) ordination of faunal assemblages sampled from 4 treatments. Abundances are fourth root transformed. a)-b) Comparison of treatments between sites at 28/06/01 and 12/07/01. c)-d) Comparison of treatments through time at Shelly Point and Jibbon. Filled symbols = site 1 (a-b) or time (c-d) ◆ = Mimic; ■ = Natural Urchin; ▲= Control; ● = Rock. Open symbols = site 2 (a-b) or time 2 (c-d) ◊= Mimic; □ = Natural Urchin; Δ = Control; ○= rock. Symbols represent site or time averages (n=6).

Comparison of treatments through time and between sites

A comparison was made to see if individual treatments varied either over time or between sites. nMDS plots showed that the assemblages within each treatment type was different between sites at both times of sampling (Fig 5 a-b). The treatments at Shelly Point at time 2 were more similar to each other than the treatments at Jibbon for the same sampling time (Fig 5b). Differences were also seen for each treatment through time. For example, the natural urchin treatment did not change much through time at Shelly Point but it changed markedly through time at Jibbon (Fig 5c-d).

ANOSIM confirmed that the differences observed from nMDS were in fact significant. Mimic and rock treatments were not significantly different through time in any site (Table 5). The urchin treatment was significantly different through time at both sites (Table 5). The control treatment varied through time but only at one site (Shelly Point) (Table 5). The mimic, control and urchin treatments were each significantly different between sites at each sampling time (Table 5). The rock treatment only varied between sites at the second sampling time (Table 5).

Table 5. The top five species contributing most towards dissimilarity between treatments at each sampling time and site, 1-5 = highest to lowest).

	Mimic & Urchin		Mimic & Rock		Urchin & Control		Urchin & Rock	
Time 1 (28/06/01)	**Site 1**	**Site 2**	**Site 1**	**Site 2**	**Site 1**	**Site 2**	**Site 1**	**Site 2**
Amphipods	2	2	1	1		3		
Amphithalamus incidatus	4				2			
Anabathron contabulatum					3			
Clanculus brunneus		3		3		1		
Clanculus clangulus		5	5		5	4	3	
Clarkoma pulchra	3						1	
Cocozeliana granaria			2				4	
Foraminifera			4		1	5		5
Opiactis resiliens				4				4
Pagurus lacertosus	1	1				2		2
Patelloida mufria		4		2				
Pisinna sp.								
Polychaete worms			3		4		2	1
Rissoina fasciata				5				3
Unknown gastropod # 6	5						5	
	Mimic & Urchin		**Mimic & Rock**		**Urchin & Control**		**Urchin & Rock**	
Time 2 (12/07/01)	**Site 1**	**Site 2**	**Site 1**	**Site 2**	**Site 1**	**Site 2**	**Site 1**	**Site 2**
Agnewia tritoniformis						3		
Amphipods		3		1				1
Amphithalamus incidatus				5				
Anabathron contabulatum							4	
Athanas granti						4		4
Clanculus brunneus				4		2		
Foraminifera	2	1	3					2
Heliocidaris eurythrogramma			2		5			
Ostracod # 1							1	
Ostracod # 2			5				3	
Pagurus lacertosus	3			3	3	1		3
Patelloida mufria						5		
Pisinna sp.					1		2	
Polychaete worms		5	1	2				
Rissoina fasciata	5							5
Tricolia sp.		4						
Unknown gastropod # 3		2						
Unknown gastropod # 4	4		4		4			
Unknown gastropod # 5	1				2		5	

SIMPER analyses identified the percentage dissimilarity within treatment types over time or in different sites with all treatment comparisons being over 50% dissimilar (Table 5). Species contributing most to the differences of assemblages varied according to the whether the treatment was observed over time or between sites (Table 6).

Table 6. The top five species contributing most towards dissimilarity within each treatment through time and between sites, 1-5 = highest to lowest).

	Mimic		Urchin		Control		Rock	
Through Time	Site 1	Site 2	Site 1	Site 2	Site 1	Site 2	Site 1	Site 2
Agnewia tritoniformis						4		
Amphipods	1	1		2	4			3
Amphithalamus incidatus					2		2	
Athanas granti				3				
Australium tentoriformis	3							
Clanculus brunneus		4				3		
Clanculus clangulus			2			1		
Cumacean							5	
Foraminifera	4	2	5	4				2
Gena impertusa	5							
Haliotis coccoradiata				5	3	5		
Heliocidaris eurythrogramma	2		4					
Opiactis resiliens								4
Ostracod # 2							3	
Pagurus lacertosus				1	5	2		
Pisinna sp.								
Polychaete Worms								1
Rissoina fasciata							1	5
Tricolia sp.		5						
Unknown gastropod # 3		3						
Unknown gastropod # 4			3				4	
Unknown gastropod # 5			1					
Unknown gastropod # 6					1			
	Mimic		Urchin		Control		Rock	
Between Sites	Time 1	Time 2	Time 1	Time 2	Time 1	Time 2	Time 1	Time 2
Agnewia tritoniformis	4							5
Amphipods					2			
Amphithalamus incidatus			4	4		4		
Anabathron contabulatum				3				
Cacozeliana granaria							4	
Clanculus brunneus	2				4	5		
Clanculus clangulus	3		5					
Clarkoma pulchra			2		5	2		
Foraminifera		3				3	2	
Gena impertusa		5						
Heliocidaris eurythrogramma		1						
Ostracod # 1								2
Ostracod # 2								3
Pagurus lacertosus	5	2	1	1	1	1	5	
Patelloida mufria	1							
Pisinna sp.				2				
Polychaete worms							1	1
Rissoina fasciata							3	
Unknown gastropod # 4								4
Unknown gastropod # 5				5				
Unknown gastropod # 6		4	3		3			

Discussion

The abundance of individual taxa were compared between treatments of *C rodgersii*, mimic urchins, rocks and the control. Most taxa occurred under rocks as well as under *C. rodgersii*. This result indicates that the species found with *C. rodgersii* have some degree of habitat flexibility and are not restricted to just the boreholes of *C. rodgersii*. Therefore the species most commonly found under *C. rodgersii* (consistently abundant species chosen for analysis) utilise at least one other habitat (rocks) and probably other cryptic habitats on NSW rocky reefs. Most species in this study have been previously reported from other habitats on rocky reefs (Edgar, 1997; Chapman, 1999).

Food and shelter resources probably influence the habitat choices made by the fauna under *C. rodgersii*. Literature shows that food availability and or shelter from predators often drive the associations with habitat (Heck & Westone, 1977; Heck & Orth, 1980; Stoner & Livingstone, 1980; Leber, 1985; Bell & Westoby, 1986; Virnstein & Curran, 1986; Connolly, 1997). Further, complex habitats generally support higher numbers of species and individuals (Robinson & Tully, 2000) because the amount of food resources and shelters tend to increase along with complexity (Connolly, 1997). *C. rodgersii* appears to be a more complex structure than rocks and so one would expect to see higher numbers of individuals and species resulting directly from increased food and shelter opportunities, but this was not found. Species richness and the numbers of total individuals showed no significant differences between treatments but they did differ between sites. Therefore, the spiny structure of the urchin would seem not to represent a more complex habitat than rocks because similar numbers of individuals and species were found in each habitat.

When the effects of food or shelter preferences were examined for individual species by comparing treatment means, it was found that only one model (model 2: food resource) was supported by one species (the brittlestar *Clarkoma pulchra*) at one site. Another species *Pagurus lacertosus* was almost significantly different among treatments in the same manner. These species appear to occur in greater numbers under the real urchin perhaps because they gain a food resource from residing with *C. rodgersii*. However, because the model was not supported in both sites, other factors in addition to food resources would better explain the presence of these species under *C. rodgersii*. Only 3 species (the brittlestar *Clarkoma pulchra*, the chiton *Ischnochiton australis*, and the hingebeak shrimp *Rhynchocinetes serratus*) that differed between treatments, were actually higher in number under *C. rodgersii* urchins (the brittlestar was absent from under rocks). Three taxa (Amphipods, polychaetes, and the gastropod *Clanculus clangulus*) were found in higher abundances under rocks. These latter taxa would appear to be less associated with *C. rodgersii* than with rock habitats. The remaining species (the clingfish *Apasmogaster costatus*, the shrimp *Athanas granti*, the whelk *Agnewia tritonifiromis*, the chiton *Ischnochiton versicolor* and the hermit crab *Pagurus lacertosus*) showed no differences between treatments at all, which indicates that these species occur in equal abundance in any of the habitats examined in this study. However, since this study was conducted at one place (Cronulla) and over a short time interval, these species may in fact differ between the 2 habitat types at different locations and times. Additionally, the clingfish was not found under rocks and the non-significant result for this species reflects the high variation associated with the treatment means. On average, less than one individual was found for each replicate, so for clingfish *C. rodgersii* may still be an important habitat.

Multivariate analyses showed that the assemblages of *C. rodgersii* were quite different to other treatments at each time and site. Both the food resource model and the shelter preference model were supported in terms of differences between assemblages but only at certain times and sites. Broad generalisations cannot be made concerning whether food or shelter influences these assemblages because each model was not supported across all sites and times. It would appear that other factors together with food and shelter preferences shape these assemblages, which was why patterns were not consistent.

The reasons why certain species or whole assemblages are found under *C. rodgersii* are complex, and not just related to food resources or shelter preferences. Because the models tested in this experiment were not fully supported by either the individual species analysed or by patterns in assemblage structure, other explanations are needed.

C. rodgersii may be selected as a habitat sometimes based on available food resources and at other times because of the spiny structure of the urchin. It is likely that the choice of habitat is based on both food resources and shelter preferences, as for epifauna occurring in other aquatic habitats (See: Heck & Orth, 1980). These two forces (out of many other possibilities) may also be acting synergistically. If both food and shelter characteristics influence habitat choice at the same time, then this could explain why neither model was singularly supported in this experiment. An alternative model is that the urchin's borehole provides a microclimate preferred by these species that is different from that provided by rocks.

Habitat selection, is usually dynamic over space and time (Virnstein & Curran, 1986: Connolly, 1997), so it is unlikely to be easily attributed to just one influencing factor. The choice of habitat for any species may be quite flexible depending on what other processes are operating. Processes that could be influencing whether certain species are found with *C. rodgersii* include the regular disturbance caused by *C. rodgersii* foraging at night, chemical cues, competitive interactions for food or space, mating migrations or ontogenetic shifts in habitat use and predation events.

The food resource model and shelter model in this study may have been confounded by the fact that mimic urchins tended to trap drift algae in their spines after the 2-week period. This would mean that the mimic was actually providing a food source (drift algae), and possibly additional shelter, when it wasn't expected to. If any species are associated with *C. rodgersii* because they feed on drift algae scraps from urchin meals, then they may have remained in the borehole or even increased in number under the mimic urchin treatment. However, since all species analysed did not support the models widely, the confounding effect of trapped drift algae was probably not important because all species would not necessarily feed on drift algae, or shelter beneath it. The species most likely to feed on drift algae was *Athanas granti* because another shrimp of the same genus has been shown to do so underneath the sea urchin *Echinometra mathaei* (Gherardi, 1991). *Athanas granti* did not however show any differences in abundance between treatments and was found under rocks.

For a mimic urchin to be a true "copy" of a real urchin it would need to move around rather than being stationary for the two-week period. However, the fact that the mimic did not move revealed an interesting result regarding the effect of the movements of *C. rodgersii* on the assemblage of associated fauna. The assemblages under *C. rodgersii* and under the control treatment were significantly different between sites and through time, which reflects a high level of natural variation. By contrast, the rock and mimic treatments did not vary through time. Mimic and rocks differed from real urchins because real urchins are mobile whereas rocks and mimics are not. Consequently, the assemblages of non-mobile treatments remained similar through time. This result is important as it may implicate the regular movement of *C. rodgersii* as a major structuring force on the assemblages inhabiting their boreholes. The suite of taxa found with *C. rodgersii* at any one time and place may be more the result of this disturbance process than due to strong habitat choices by the associated fauna.

The mechanism of this structuring force can be understood in the following way. Many species found in this study appear to utilise other cryptic habitats (as shown by their presence under rocks). When these fauna seek shelter following movements of various distances over the reef at night (when *C. rodgersii* leaves it's borehole), their proximity to *C. rodgersii* compared to their proximity to a variety of similar cryptic habitats probably determines which habitat they will choose. Any food and specific shelter preferences of these species may therefore play a secondary

role if these species settle in the nearest available habitats at daybreak. Stoner, 1985) suggested that the composition of macrocrustaceans inhabiting macroalgae was influenced both by their food and shelter requirements and their nocturnal mobility. Taylor & Cole, 1994) also found that epifauna inhabiting different brown seaweeds in New Zealand showed low host specificity and suggested that this pattern was due to their non-selective food requirements and nocturnal movements. Taylor & Cole, 1994) suggested that where these fauna occur, depends on far they move at night and what algae are available when they resettle out of the water column. Similar movements of intertidal invertebrates at high tide results in regular changes in their distribution and abundance (Underwood & Chapman, 1996).

It was noted that some types of species tended to occur more under rocks than urchins and vice-versa. Generally, both smaller individuals and smaller species types were found under rocks. The size of an individual influences habitat choices since the effectiveness of any refuge is determined by the size of the prey in relation to it's habitat (Krebs, 1994). Perhaps different life stages are found under rocks compared to *C. rodgersii* for some species. Individuals of different species may migrate to other habitats such as *C. rodgersii* later in their lifetime as they grow larger. Such ontogenetic shifts occur with decapod crustaceans and larger more mobile species may be more capable of migrations from less favourable sites (Robinson & Tully, 2000). Further, if larger individuals are also older, then these larger individuals occurring under *C. rodgersii* may constitute the breeding proportion of the population for some species. The smaller individuals occurring under rocks may indicate that juvenile stages of these species prefer rock habitats or that rocks are areas where larvae successfully recruit. Choices of substrate by settling larvae and post-settlement processes can be determined by substrate type and this may affect community structure (Robinson & Tully, 2000). Perhaps rocks provide a more stable environment for new recruits.

Larger taxa that are very mobile (shrimps) may be better able to survive in the *C. rodgersii* habitat because the spiny structure allows them to move behind a continuous cover when urchins are close together. Rocks would be more restrictive in terms of large mobile species such as shrimp being able to move around beneath them. Species that are dorso-ventrally or laterally flattened such as chitons, amphipods and isopods may be less inhibited by the lack of space for movement beneath rocks. Likewise, polychaete worms probably prefer the rock habitat since more sediment was found in these areas than in the boreholes of urchins.

The factors that influence habitat selection in the marine environment are complex and this was shown by this experiment. Food or shelter related preferences were not clearly the only factors influencing why certain species are found together with *C. rodgersii*. Although habitat specificity was not demonstrated for most species in this study, at least 3 species (the brittlestar *Clarkoma pulchra,* the shrimp *Athanas granti* and *Apasmogaster costatus*) require further evaluation. The latter two species were not found under rocks and literature has indicated previously that brittlestars, clingfish and shrimp have been found in association with sea urchins (Gherardi, 1991; Schoppe & Werding, 1996; Goncalves *et al*, 1998; Hofrichter & Patzner, 2000). Since these species were found in typically very low abundances, research other than the quantification of their abundance in various habitats may be required to assess their relationship with *C. rodgersii*. For example, gut analyses could determine their diet and surveys of their presence in a variety of cryptic habitats through several seasons may determine their habitat specificity. Additionally other key aspects of their ecology such as breeding periods, behaviours and movements should be understood before impacts from a sea urchin fishery to these species can be ruled out.

For the remainder of the species found during this experiment it would appear that they do exist in other cryptic habitats on NSW rocky reefs and that they probably do not have an obligate association with *C. rodgersii*. However, since *C. rodgersii* is numerically abundant on NSW rocky reefs, their removal via fishing may represent a loss of valuable habitat for these species. The magnitude of this type of impact will depend on the relative availability of various microhabitats on

a reef-wide scale and what proportion of each species population is supported by the *C. rodgersii* habitat.

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APPENDIX 7: Market assessment of Australian sea urchin roe

Chemosense 4(4)

Japanese YEN for Australian Sea Urchin Roe

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Large populations of sea urchins exist on the eastern coast of Australia. This stock of wild sea urchin may be sufficient, if sustainable, to create a viable and lucrative export trade, to countries where sea urchin roe is consumed, particularly Japan, where "uni," as it is called there, is very highly valued. A field trip to Japan was undertaken to gain a better understanding of the characteristics of high value roe and how these might be obtained or enhanced in Australian stocks. A market profile for fresh sea urchin roe must be developed to refine enhancement and processing techniques. The results, summarised here, and have been reported elsewhere in more detail (Blount et al., 2001).

Despite long-standing interest in the fishing of sea urchins and their roe in Australia (Ward, 1975, James, 1990), attempts to develop an export industry of the roe or the whole urchins has met with very limited success. It is only recently that sales of Australian roe to Japan have become measurable: 3449 Kg in 1999, valued at ¥20,676,755 or AUD$333,496 (conversion rate = 62 ¥/AUD$). The largest exporter to Japan in that year was USA with over 2 million Kg. Although the Australian product was exported in low volume compared with most other countries, it fetched one of the highest prices (¥5995/Kg or AUD$97/Kg), indicating the potential Australian urchin has to compete on quality in that market.

Australian sea urchin is a valuable resource with potential to generate significant Australian foreign trade earnings, provided management of wild stocks can produce high quality product and sustainable yields.

Raw roe is the value-added form of the sea urchin product. The eggs are tiny: almost indistinguishable by eye. They form a smooth mass, contained in five leaf-shaped egg sacs which hold their integrity when carefully removed from the animal. These sacs are sold as a highly prized delicacy, "uni", throughout Japan, with individual servings of raw roe selling in top restaurants for ¥5 000 per 20 gm serve (AUD$4000 per Kg).

Australia can capture part of this value-adding process by supplying raw roe directly to the Japanese market.

Our fisheries authorities need to know the value of the wild stocks, the kinds and quantities of urchin contained therein, and how to manage these stocks in order to achieve a viable and sustainable sea urchin industry.

This study was undertaken in the field in Japan in November, 2000, in order to gain an understanding of the market forces that might operate on Australian sea urchin roe exports particularly from the view-point of what consumer drivers of acceptance of uni must be met by Australian exporters.
The field excursion also provided an opportunity to see Japanese uni processing and sea urchin aquaculture operations, as well as to meet people involved in importing sea urchin and see the methods for distribution of uni at Tsukiji market in Tokyo.

Demand

In Japan, sea urchin roe is regarded, along with tuna, lobster and abalone, as a premium seafood product. Consequently, there is a strong demand for sea urchin roe throughout Japan. Traditionally, sea urchin roe has been eaten mostly at sushi bars and restaurants. Japanese consumers generally prefer local products over imports, and this is quite evident at the premium sushi restaurants in Tokyo, who serve, it is said, only Japanese sea urchin roe. Observations in Osaka, Tokyo and Hakodate (on the northern island of Hokkaido) confirmed that uni is popular and ubiquitous in sushi restaurants, of which there are many thousands in Japan. Expert opinion is firm that demand for uni in the

restaurant trade is as strong as it ever has been. General demand can be expected to vary with disposable income and product availability.

More recently, sea urchin roe distribution has widened, becoming available to a new market of Japanese consumers: supermarket shoppers.

Hokkaido: The centre of Japanese uni industry

Japanese consumers can now obtain uni at many suburban supermarkets, where prices for food to be consumed at home are expected to be lower than for food prepared in restaurants and sushi bars. This has created an outlet for lower grade imported sea urchin roe being sold on supermarket shelves. Demand for uni throughout Japan is therefore likely to rise, as supermarkets are relatively new to Japan and with relaxation in laws controlling retail shop sizes being gradually relaxed throughout the country (Bell et al., 1992). The high position of uni in Japanese cuisine will also drive this increase in demand for supermarket-bought product.

There are regional differences in consumer demand for sea urchin roe, and the Tokyo and Osaka markets in general demand a higher quality product than the regions, as indeed they do for most goods in Japan. These two cities are the portals for entry of imported sea urchin and centres of maximum demand by consumers of the

Whole uni in cool store awaiting processing

product.

Although the economy of Japan is presently experiencing low growth, Tsukiji market stakeholders gave assurance that this has not affected demand for uni.

It is likely that the demand for both high and low grade product will increase, due to a continual decline in production of local high grade material, and an expanding supermarket trade. Australian producers can aim to send their product into either or both the Tokyo and Osaka markets.

Prices of Australian product should benefit from competitive pressures originating from the various sources of demand in Japan.

Supply

There are two crucial factors affecting supply of sea urchin roe Imported into Japan: Firstly how much acceptable sea urchin roe a country can produce, and secondly, whether the time of harvest (normally outside the spawning period) coincides with demand, affected by local supply and competitors' ability to supply uni to market.

USA, Mexico, Chile and Canada provide the bulk of imported sea urchin roe to Japan by volume, although substantial amounts are also imported from China and Korea. Sea urchins from most supplying countries have a harvesting period complementary to most Japanese species, but price advantage is not necessarily accrued, as most imports generally fail to compete with Japanese sea urchins for the premium end of the market.

Diminishing Japanese stocks reached a point recently when it was estimated that less than half of the total volume of sea urchin roe consumed in Japan is produced locally. Further decreases are expected due to declines in local fisheries. Reasons for the decline are unclear, but overfishing, pollution, El Nino and other environmental factors are of some concern. Australia needs to study the experience of the Japanese in this regard, in case lessons may be applied to Australian sea urchin resource management.

The reputation of Japanese sea urchin as the best available and its price premium sustains the pressure on local supply. Japanese processors interviewed believe that the quality of their labour in fishing, handling and processing the product, and some secret knowledge concerning temperatures and chemical composition of the process-baths, allows them to produce products of a much higher quality than imports.

Handling the product optimally from the water to the customer is in every Australian sea urchin exporter's best interest. The importance of handling, processing, packing for best presentation, and cold-chain effectiveness is even greater than country of origin of the sea urchin. This is evidenced by reports to the team by Hokkaido processors who told of quantities of whole urchin arriving from the Boston, USA, being processed in Hokkaido and the roe sold under Japanese label.

So, sea urchin products that are produced in Japan generally sell at a premium regardless of the source of raw material, and imported products processed under a local brand name, can be sold as Japanese product, at the usual premium.

Quality and Price

Japanese culture and tradition underpins the variables determining the perceived quality of sea urchin roe. Because Japanese consumers prefer products produced in Japan, the highest prices paid for sea urchin roe are from local species. For example, *Bafun* uni can easily fetch five times the average price of imported sea urchin roe. The message to overseas producers is clearly to process sea urchin roe so it looks as similar to Japanese products as possible. This is possible for some species (eg *S. droebachiensis,* East Coast of USA), which are sometimes shipped whole to Japan and processed and sold under a local brand name, so they appear to the consumer as local product. One hundred grams of *S. droebachiensis,* processed in Hokkaido, can fetch up to ¥12,000 (AUD$193) at auction.
However, the majority of imported sea urchin roe is much larger, and is impossible to be made to look like the roe of Japanese sea urchins. These products will always be down-graded, as they will be recognised as foreign.

Variability between countries in the quality of sea urchin roe leads to a range of prices paid for imported product.

Colour is widely agreed as a factor in determining the value of the roe. Yellow, with gradation of colours to orange is highly regarded as is a yellow ochre colour that is the basis for the Japanese word "bafun" applied to a highly valued species of local sea urchin, meaning horse droppings.

Colour, size, and texture must be consistent among the trays in a batch. The study team watched uni packers plying their art with great dexterity. The less perfect specimens are used as "filler" and the most beautiful ones are layered over them. The colours are varied so that the overall impression is of an average, good coloured set, even though the individual sacs do have many different shades of colour. Reputation of the supplier depends on consistently good presentation of the product.

Japanese we interviewed agreed that freshness is determined from the look of the uni: they must be firmly integrated and not oozing or dry.

Very little is known about what, and to what degree, flavour and mouthfeel (texture) attributes play a part in grading the quality of uni. Buyers at auction at the Tsukiji market have an opportunity to inspect and request a taste of the uni on offer. Cuisine experts reported to the team that sweetness is very important. The uni must "melt slowly on the tongue like a good European chocolate, infusing the mouth with its many delicate flavours," said a Tokyo chef interviewed.

Bitter roe (such as roe from Chile, we were told) fetches a low price, even if the colour is reasonable, indicating that bitterness is a negative driver of product acceptance. There is not a lot of sourness or saltiness in uni, and the roe are often eaten with a sour and salty sauce, such as soy sauce.

Texture in the mouth is also important: the uni must not be rough, but creamy, yet the feeling of the uni disintegrating on the tongue and releasing flavours is linked to a perceivable amount of granular texture.

It should be possible for Australian expert tasters to learn to rate the taste of uni on the same criteria once these can be quantified. An expert sensory panel could be used to monitor quality of the Australian product. At this stage of the development of the Australian industry it would be advisable to monitor taste using trained expert sensory evaluation rather than by encouraging the development of "fashion gurus" as are found in the wine industry. Feedback from market price obtained will be the feedback upon which to base the ultimate sensory benchmark.

Of all the attributes that are used to judge quality, colour and taste (flavour) appear to be the most important. The industry needs clear sensory guidelines for colour and flavour.

NSW and Eastern Victorian Sea Urchin

The sea urchin industry in NSW (and eastern Victoria) involves the harvest of three species. The red sea urchin (*H. tuburculata*) and green/white sea urchin (*H. erythrogramma*) have the best taste, but the colour of the roe in these species is variable, and the harvestable quantity of these species is small in comparison to the third species, the purple urchin *(C. rodgersii*). Despite a somewhat bitter taste in the roe immediately after the spawning period, purple sea urchin can be caught in vast quantities and, if harvested from the right area, the roe can be of a consistent yellow colour. There is also a window in the Japanese market during the harvest period for purple sea urchin (December - June), as the US and Canadian sea urchin spawn at this time. Further research is needed on purple urchin to determine if selective harvesting or other variables might maximise the desirable qualities (such as sweet taste).

Red sea urchin can be harvested all year round, due a protracted spawning period, and the green/white urchin are best harvested from July - December. These complementary spawning periods allow the possibility of a continuous supply of roe from NSW and Eastern Victoria to Japan, and year round employment for divers and processors.

Hatchery and Re-seeding

Aquaculture is used in Japan to re-stock locally depleted areas of sea urchin. In fact, the sea urchin industry in Japan is totally dependent upon a restocking program. Other countries are also finding that their wild fisheries are not sustainable at present levels of harvesting, without some form of enhancement. In Japan, sea urchin are grown for one year in tanks and released to areas where they are left to grow for another year. At two or three months prior to harvesting, they are given a seaweed supplement

Final grading and packing of Japanese uni

to feed upon. This apparently, increases the yield, colour and taste to a desirable level. Many advances are presently being made in the field of sea urchin aquaculture around the world. Australia is no exception, and presently a research program, based in South Australia, is on the way to developing a commercial enhancement technique for cultured green/white sea urchin.

Urchin hatchlings are in themselves a valuable product. The specimens measuring 1-2 cm in diameter produced at the hatchery visited in Hokkaido are sold to the local fishing industry for 10 cents each. A subsidiary industry is possible from the direct sales of the small urchin to areas needing reseeding or as another export product to producer countries.

Australia stands to benefit from learning more of the pioneering work of Japanese hatchery and re-seeding operations.

Future use of chemosensory techniques

The importance of sensory quality of "uni" was found to be paramount. Australian product cannot succeed in export markets if it does not meet expectations of the palate of consumers in those markets. This study found reference to flavour evaluations from the processing and

grading steps, to the market agents and buyers and throughout the market's route to Japanese consumers of uni.

In summary, expert sensory panels can be used as a resource management tool and as a market development tool. The immediate need is to develop the techniques needed for an expert sensory panel to serve resource management issues and overall enhancement of roe quality in sea urchins from wild populations.

An expert panel could be located in either country: Japan or Australia. The key perceptual judgements of the experts are identified by correlation of their assessments of uni with the acceptance score of the Japanese consumers. Once an Australian panel is trained to assess the key characteristics it can be put to work regularly in pursuit of the aim of enhancement of roe quality in sea urchins from wild populations. It would regularly assess urchins from various regions etc, so that production variables can be related to the key product quality attributes. The knowledge thus gained would be used to guide fishery management practices so that the resources remain sustainable and of maximal economic value.

The local uni industry could then be educated in the establishment of its own panels for quality assurance, production improvement and market development purposes. These panels would be independent of the research panel.

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APPENDIX 8:

Effects of removal of urchins from crevices on associated invertebrate assemblages

M.G. Chapman and A.J. Underwood

(*Report by the Centre for Research on Ecological Impacts of Coastal Cities, University of Sydney*)

EFFECTS OF REMOVAL OF URCHINS FROM CREVICES ON ASSOCIATED INVERTEBRATE ASSEMBLAGES

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Centre for Research on Ecological Impacts of Coastal Cities
Marine Ecology Laboratories (A11), University of Sydney
NSW 2006

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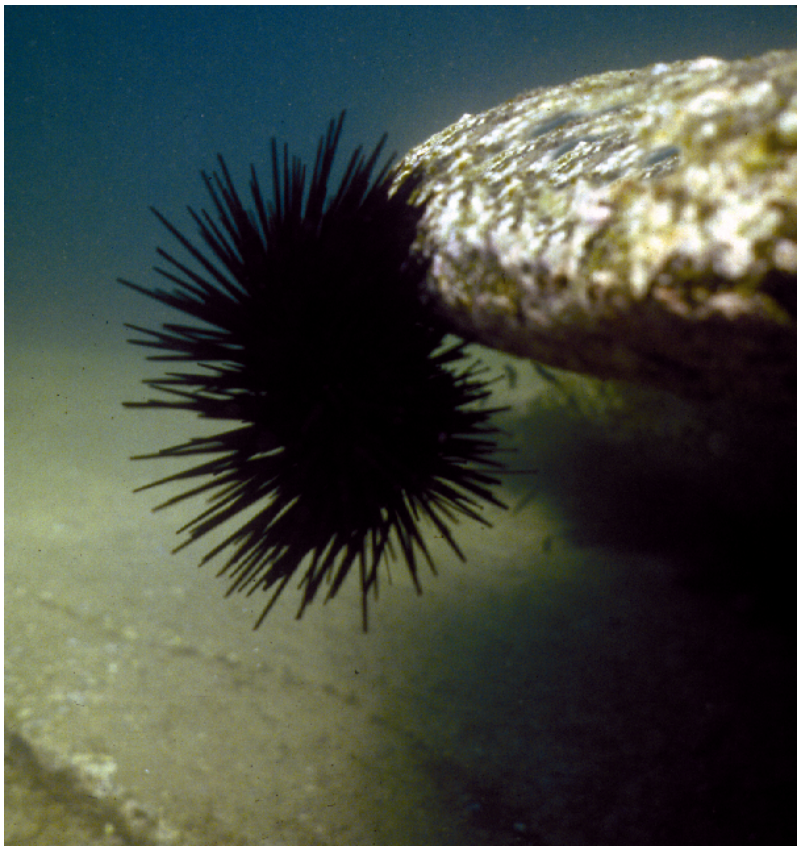

INTRODUCTION

With widespread concern about "ecologically sustainable development", there is an increasing need to manage resource-extraction, including commercial fisheries, in a manner that conserves marine biodiversity of target and non-targeted species and the structure and function of natural habitats (Ludwig *et al*., 1993; Botford *et al*., 1997; Hoel, 2003). In addition, because the 1992 United Nations Convention on Biodiversity requires impact assessment of projects likely to adversely affect biodiversity, there is a moral obligation to consider commercial fishing in this context. The consideration of impacts on marine biodiversity and habitats is an important component in the current development of Fisheries Management Strategies and Environmental Impact Assessments for the various commercial fisheries in New South Wales.

Nevertheless, until relatively recently, most concern about effects of commercially fishing was focussed on the widespread and destructive effects of bottom-trawling on benthic habitats and associated non-targeted species (e.g. Collie *et al*., 2000; Lindegarth *et al*., 2000; Cryer *et al*., 2002). A number of studies on the volumes and types of by-catch (particularly those that are discarded) taken by commercial fisheries has led to attempts to changed fishing practices in an attempt to reduce impact on non-targeted fish (e.g. Kennelly, 1995; Hall, 1999; Gray, 2002). Apart from effects of hand-gathering of intertidal species in Chile (Moreno *et al*., 1984; Castilla and Duran, 1985), South Africa (Lasiak and Field, 1994; Lasiak, 1999) and, to a smaller extent, Australia (Kingsford *et al*., 1991; Keough *et al*., 1993), there has generally been less concern about effects of hand-gathered fisheries on targeted or non-targeted species. This is particularly so for subtidal gathering, except for fishing of abalone, which are very widely collected and considered extremely vulnerable in some areas (e.g. Tegner, 1993; Tegner et al., 1996).

Sea urchins are also commercially collected from subtidal habitats in many parts of the world (Tegner and Dayton, 1977; Smith and Berkes, 1991; King *et al*., 1994; Watson and Ormond, 1994). Most concern about decreases in abundances of urchins by commercial fishing has centred around their perceived pivotal role in maintaining urchin "barren-grounds" and, simultaneously, controlling the extent of areas of large foliose algae, including kelp-beds. There have been many experimental studies on effects of changing densities of sea urchins on cover of algae (e.g. Fletcher, 1987; Watanabe and Harold, 1991; Konan and Estes, 2003, among many others). There have been rather fewer studies on effects of changing densities of urchins on other associated invertebrates (e.g. limpets, Fletcher and Underwood, 1987; Andrew and Underwood, 1989), mediated through either direct interactions among urchins and other

grazers, or indirectly via effects that changing densities of urchins have on algal cover. Fletcher and Underwood (1987) showed decreased densities of the limpets, *Patelloida mufria, Cellana tramoserica* and *Patelloida alticostata*, concomitant with an increase in cover of foliose algae and a decrease in cover of crustose algae, when densities of the large urchin, *Centrostephanus rodgersii* were decreased experimentally in shallow, subtidal barren grounds off Cape Banks, New South Wales, Australia. Following a mass mortality of the urchins, densities of limpets increased and then subsequently declined with increasing cover of foliose algae on barren grounds (Andrew 1991). Other studies have not, however, demonstrated any relationships between abundances of sea urchins and other invertebrate grazers (e.g. Scheibling and Raymond, 1990; Watanabe and Harold, 1991).

Many sea urchins live in crevices and other cryptic habitats, such as underneath boulders, only emerging to feed at night, or less frequently. This behaviour has been attributed to avoidance of predators (e.g. Nelson and Vance, 1979) or greater survival in sheltered habitats during rough seas (Konan and Estes, 2003). Their sit-and-wait feeding strategy, which allows them to subsist on drift algae (Lowry and Pearse, 1973), means that they do not have to forage continually outside crevices in order to survive.

The large sea urchin, *C. rodgersii*, has been shown to be positively associated with crevices along the coast of New South Wales (Andrew and Underwood, 1989). Other invertebrates are also abundant in crevices used by urchins. For example, Andrew and Underwood (1992) showed that densities of the sea urchin, *C. rodgersii* and abalone (*Haliotis rubra*) were negatively associated at scales from individual crevices to kilometres. Crevices, boulders and other cryptic habitats are also used by a variety of other mobile invertebrates (e.g. Chapman and Underwood, 1996; Robinson and Tully, 2000). Urchins and abalone feed on drift algae and it has been suggested that they may compete for suitable crevices (Lowry and Pearse, 1979).

There are, however, also strong positive associations between some mobile invertebrates and urchins (e.g. the chiton, *Ischnochiton australis*). These two species are strongly positively correlated among boulders and among sites on individual boulders in intertidal/shallow subtidal boulder fields in New South Wales (unpubl. data). They appear to be associated in small depressions in the boulders, although this pattern has not been quantitatively measured. There may be many causes for positive associations between adult urchins and other smaller grazers. In an experimental caging experiment, Tegner and Dayton (1977) showed that juvenile *Strongylocentrotus franciscanus* almost exclusively recruited under the spines of conspecific adults, whereas *C. purpuratus* were more cosmopolitan in sites of recruitment. This was due to

a strong behavioural response on the part of the juveniles, which actively moved towards and stayed in close association with the adults. Tegner and Dayton (1977) also reported small abalone and ophiuroids living under the spines of adult urchins.

The development of a commercial fishery of *C. rodgersii* along the coast of New South Wales therefore has the potential to alter habitat for a suite of other small species. This may be an indirect effect, e.g. by altering composition and cover of algae, thus strongly changing availability of suitable habitat for many taxa. It may also occur directly by removing potential habitat (the urchins, themselves). The former effect is most likely to be perceived on the barren grounds on which the urchins may graze. The latter effect is more likely to occur in the crevices, where the urchins shelter.

This pilot study examined effects of removal of urchins (simulating commercial fishing, but at a much smaller scale) on the diversity of other mobile invertebrates that occupy the same crevices as the urchins.

MATERIALS AND METHODS

Field methods

After preliminary work at Bare Island, Cape Banks and Inscription Point, trialling methods of removing urchins, maintaining the treatments, attaching exclusion cages and sampling the associated organisms, experiments were set up at Cape Banks, Port Kembla, Swansea and Jervis Bay, New South Wales. The cages at Port Kembla were destroyed in rough weather and only one site, in which cages could be attached, could be found at Swansea. Therefore, the experiment was restricted to 2 sites at each of Cape Banks and Jervis Bay (Figure 1). Each site had relatively large abundances of *C. rodgersii* in crevices in sandstone barren grounds, was easily accessible by boat and the hardness of the rock allowed attachment of cages to maintain urchin-free crevices. This was done to mimic commercial fishing which would have removed urchins over very large areas and where, therefore, crevices would not be rapidly recolonized by adult urchins.

Sites were selected in relatively shallow water, 2 – 7 m deep, with enough suitable crevices for independent replication of all treatments within an area of approximately 1000 m^2. Crevices were considered suitable if they were no more than 2 m deep from the mouth to the base, had relatively large densities of *C. rodgersii* and had a mouth wide enough to remove (or otherwise disturb) urchins as required and to insert the suction pump to sample the invertebrates. All crevices were at least 2 m long and were separated by at least 1 m.

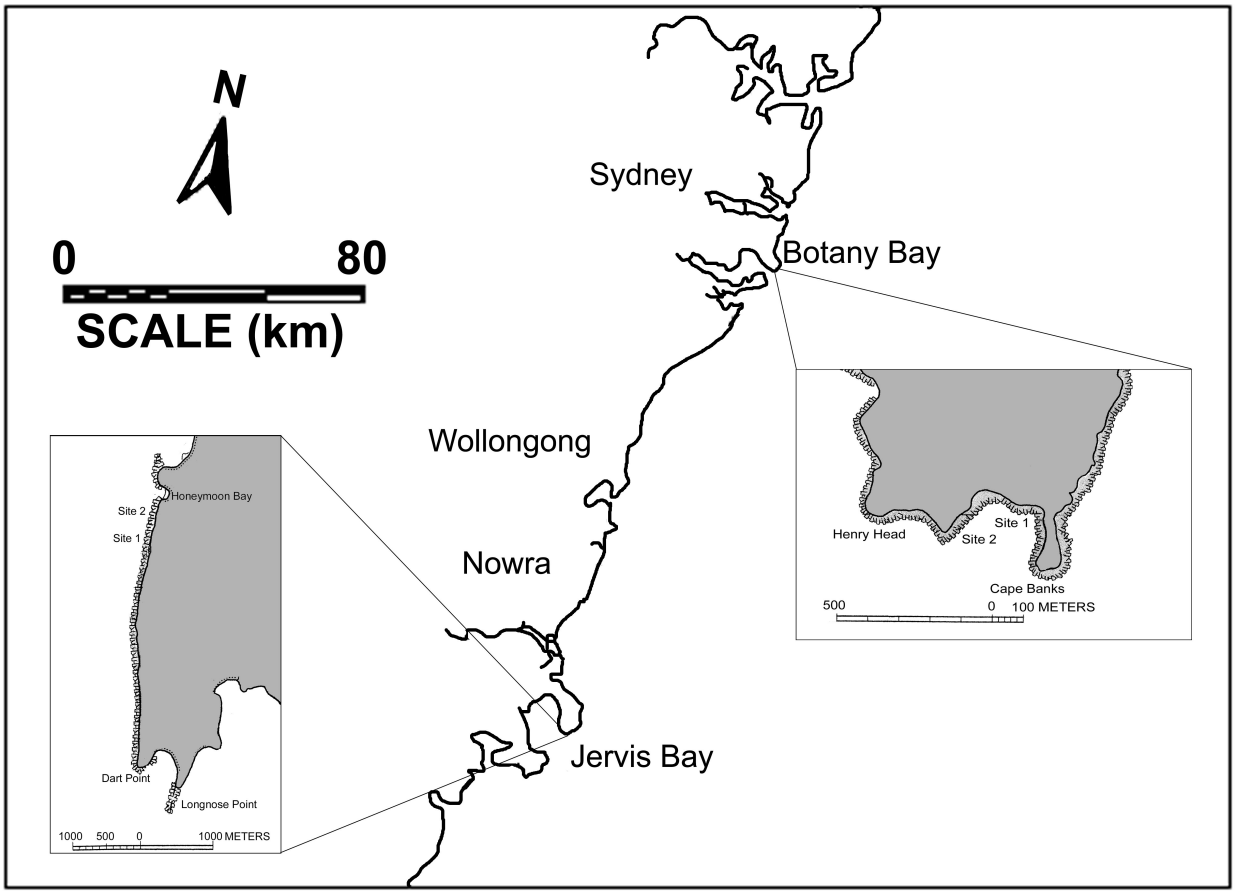

Figure 1. The location of the study sites at Cape Banks (Botany Bay) and Honeymoon Bay (Jervis Bay).

At Cape Banks, a smaller section of a long crevice was used for some treatments because there were not enough separate crevices. In this case, there was at least 5 m between treatments in the same crevice. Crevices were identified, marked with a numbered tag nailed onto the top of the crevice and then randomly allocated to one of the five treatments. The positions of all crevices were then mapped.

Two experimental treatments were necessary to test the hypothesis: (T1) no disturbance of *C. rodgersii* and (T2) removal of all *C. rodgersii* from the crevices. Because it was not possible to maintain Treatment 2 without urchins rapidly recolonizing the crevices from adjacent areas, cages were needed to keep them out. Therefore, three procedural controls were also needed: (C1) a control for fencing (urchins removed, but a partial cage used so that urchins could move into the crevice), (C2) a control for disturbance (urchins removed and then replaced without any fencing) and (C3) control for fencing and disturbance (partial cage used, with the urchins replaced in the crevice).

Cages were made of Duramesh© (50 mm x 50 mm plastic-coated mesh). Full cages completely covered just over 2 m of a crevice with mesh and was attached to nails in the rock

using a tying tool. The sides of the crevice were caged using mesh cut to height, forced into the crevice and attached to the main cage. Partial cages were constructed as above leaving a one metre opening at the front of the crevice to facilitate immigration and emigration of urchins.

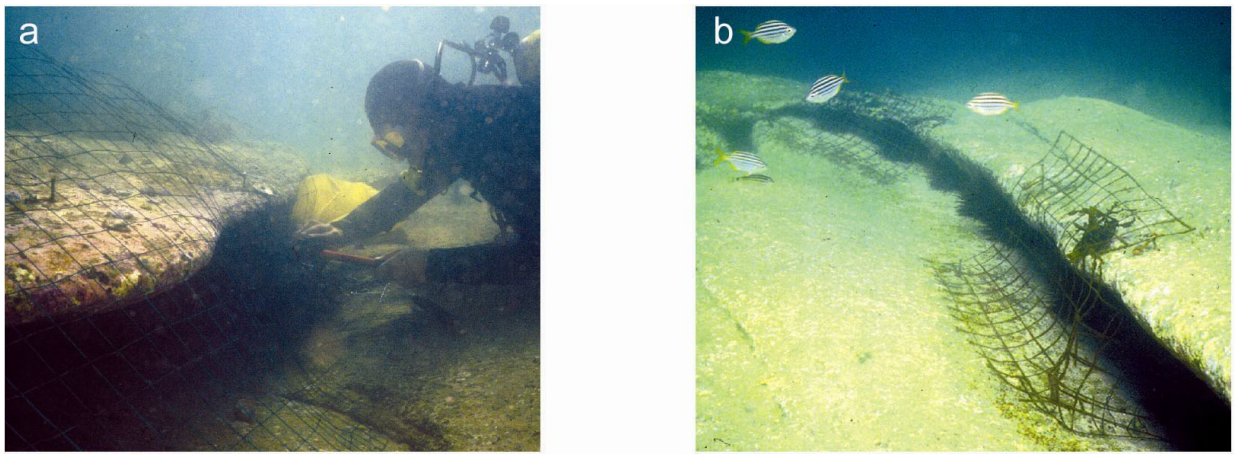

Figure 2. (a) Diver attaching a cage to the substratum; (b) a control for caging (C1)

Urchins were removed by reaching behind them with a two pronged, L–shaped rod and pulling them out of the crevice. This mimicked commercial fishing.

Two months after the treatments were set up (January, 2002 for Jervis Bay and July, 2002 for Cape Banks), the invertebrates living in the crevices were sampled. A solution of 20 % Clove oil in seawater was squirted into each crevice (Ackerman and Bellwood, 2002). This anaesthetised the invertebrates, causing them to loosen their "grip" on the substratum. A metal scraper (10 mm blade) was also used to dislodge any limpets and chitons. The assemblage was then sucked into a venturi-suction sampler (similar to that described in McShane and Smith (1988)), using surface-supplied air from a compressor into a 500 µm mesh bag.

Laboratory methods

All material was preserved in buffered 7% formalin before being sorted. The material was sieved through 8 mm, 1 mm and 0.5 mm sieves to divide the assemblage into three size-fractions. The 8 mm fraction was sorted completely with no magnification. It mainly consisted of large shell fragments. The fauna in this fraction was sparse and mainly included large gastropods, such as *Australium tentoriforme* and *Calliostoma speciosum* and chitons.

The 1 mm fraction was sorted under a magnifying lamp and the 0.5 mm fraction was sorted under a dissecting microscope. Due to the large amount of coarse sand and shell-grit in samples from Jervis Bay, it was necessary to subsample when it was estimated that any

component would take > 90 minutes to sort (i.e. 3 hrs total for the 1 mm and 0.5 mm components). Subsampling consisted of measuring the overall volume of the sample in a graduated beaker and then sorting small portions taken haphazardly from the beaker for a maximum of 90 minutes. The remaining volume gave a measure of the proportion that was sorted.

RESULTS

Numbers of urchins

There were significantly fewer *C. rodgersii* in the caged treatments than in all other treatments, which did not show significant differences (Table 1; Figure 3). Thus, the treatments were substantially maintained throughout the experiments.

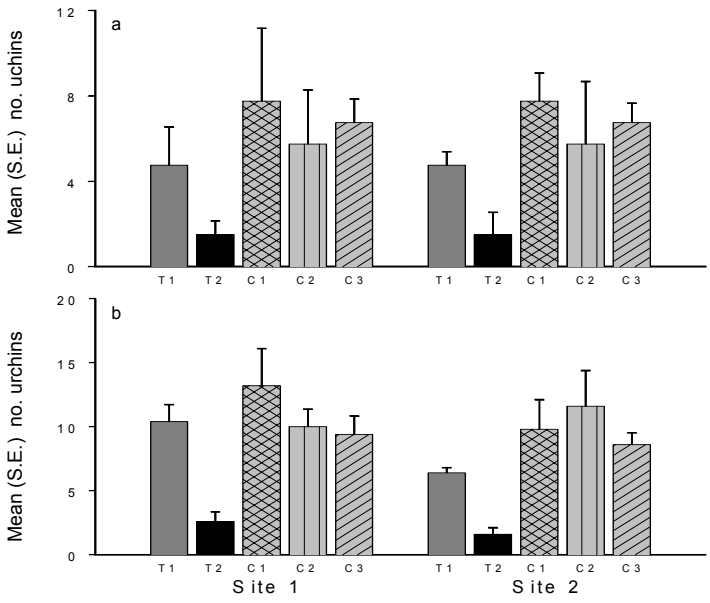

Figure 3. Mean numbers of urchins per treatment at (a) Jervis Bay and (b) Cape Banks; T1 no disturbance; T2 urchins removed; C1 urchins removed, partial cage; C2 urchins disturbed; C3 partial cage, urchins replaced; *n* = 4.

Table 1. Analyses of the mean number of urchins per crevice at the end of each experiment; *n* = 4. In this and subsequent Tables, * = *P* < 0.05; ** = *P* < 0.01; *** = *P* < 0.001.

		Jervis Bay		Cape Banks	
Source of variation	**df**	**MS**	***F***	**MS**	***F***
Site = S	1	67.6	4.74*	24.0	1.49
Treatment = T	4	55.7	8.06*	126.8	7.53**
S x T	4	6.9	0.49	21.7	1.34
Residual	30	14.3		16.2	

Cape Banks, test for Treatment after pooling Residual and S x T interaction.

Associated assemblages

Taxa were sorted to different resolution according to their diversity and abundance in the different sites and according to local taxonomic knowledge. The numbers of taxa found across all treatments in both sites are summarised in Table 1. Most taxa were sorted to coarse taxonomic level, but the gastropods, chitons and limpets were sorted to species at Cape Banks, where they were more numerous, diverse and where strong treatment-effects were apparent. A subset of these gastropods were sorted further in the two experimental treatments (T1 and T2) at Jervis Bay, to get measures of abundance to test the hypothesis that any differences between caged and control treatments were similar across bays. Limpets and gastropods were treated as two different "taxa" because the coiled gastropods were not as firmly attached to the surface and were therefore probably more consistently sampled.

Table 2. The taxonomic resolution for the different invertebrate groups. Limpets, chitons and other gastropods were sorted to species at Cape Banks; gastropods were sorted to species at Jervis Bay; +/- indicates presnce/absence; numbers indicate number of species identified for particular taxa.

Taxa	Jervis Bay	Cape Banks
Polychaeta	+	+
Other worms	+	+
Mite	+	-
Fish	+	+
Pycnogonida	+	-
Hermit Crab	+	+
Hinge-back Shrimp	+	+
Other Crustaceans	+	+
Ophiuroidea	+	+
Bivalve	+	+
Chiton	+	7
Limpets	+	13
Gastropoda	58	95

The first set of analyses examined differences in assemblages among all treatments in each site with taxa sorted to coarse taxonomic resolution (i.e. 13 taxa). Data were analysed using npMANOVA on Bray-Curtis dissimilarity measures calculated for untransformed data, with permutation of the raw data (Anderson, 2001). Locations were analysed separately (Treatments, fixed, 5 levels; Sites, random, 2 levels; $n = 4$).

There were no significant differences among treatments at Jervis Bay, nor in Site 1 at Cape Banks (Table 2), but the assemblage in the fully caged treatment (T2) in Site 2 at Cape Banks differed significantly from all other treatments, which had similar fauna. Patterns among treatments are shown in Figure 4, for each site separately.

Table 3. Analyses of the assemblages in each site at Jervis Bay and Cape Banks

		Jervis Bay		Cape Banks	
Source of variation	**df**	***F***	***P***	***F***	***P***
Site = S	1	8.82	< 0.01	8.05	< 0.01
Treatment = T	4	0.57	> 0.05	1.30	> 0.05
S x T	4	1.49	> 0.05	2.46	< 0.05

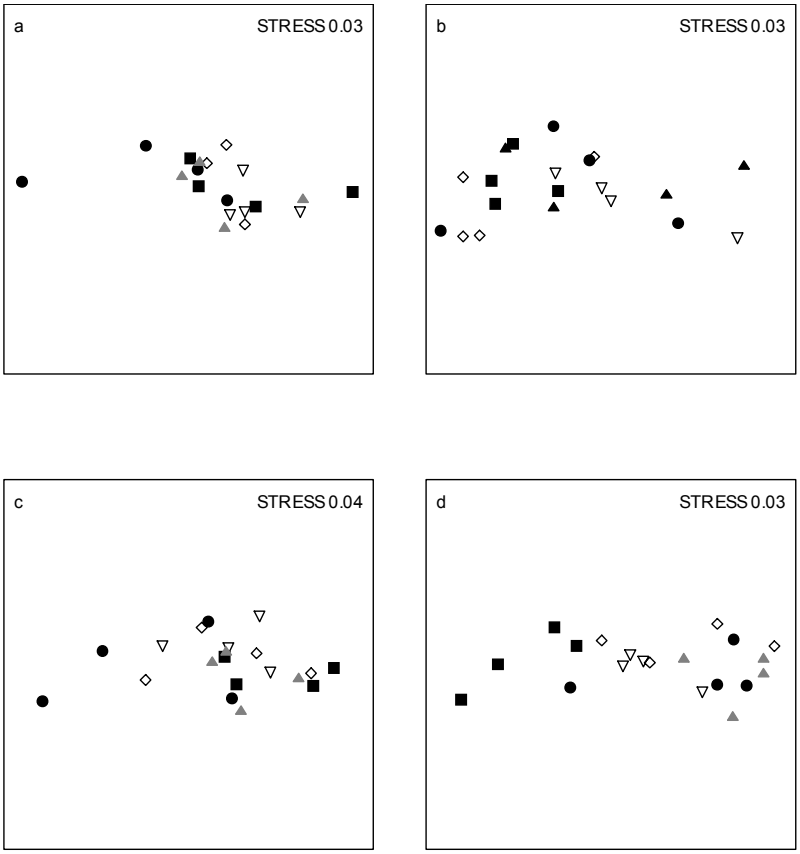

Figure 4. nMDS plots for all treatments in (a) Site 1, Jervis Bay, (b) Site 2, Jervis Bay, (c) Site 1, Cape Banks, (d) Site 2, Cape Banks; ● - T1 (undisturbed); ■ - T2 (full cage); ◇ - C1 (partial cage); ∇ - C2 (disturbance); △ - C3 (partial cage and disturbance); *n* = 4.

At Site 2 at Cape Banks (the only site that showed a significant difference between the fully caged treatments and all others), the taxa that contributed most to dissimilarity between the two experimental treatments were measured using a procedure based on SIMPER (PRIMER; Clarke, 1993). Gastropods contributed 55 % and 51 % to differences from replicate to replicate within T1 (control) and T2 (full cage), respectively and 41 % to differences between the treatments. Crustaceans contributed 48 % and 53 % to dissimilarity within T1 and T2, respectively and 50 % to dissimilarity between these two treatments.

Therefore, the abundances of gastropods and of crustaceans (with no further subdivision at this stage) were analysed across both sites at Cape Banks. Although there were

no significant differences in the abundances in either site for any analysis (Table 4), there were smaller abundances of each taxon in the fully caged treatment at each site. The various control treatments had very variable abundances and there were no consistent patterns among the controls, or between them and the two experimental treatments at any site.

Table 4. Analyses of the mean number of crustaceans (transformed to natural logarithms due to extremely heterogeneous variances) and gastropods per crevice in the two sites at Cape Banks; *n* = 4.

		Crustaceans		Gastropods	
Source of variation	**df**	**MS**	***F***	**MS**	***F***
Site = S	1	2.18	3.22	50268	20.00***
Treatment = T	4	3.45	2.29	9562	1.59
S x T	4	1.50	2.22	5997	2.39
Residual	30	0.68		2514	

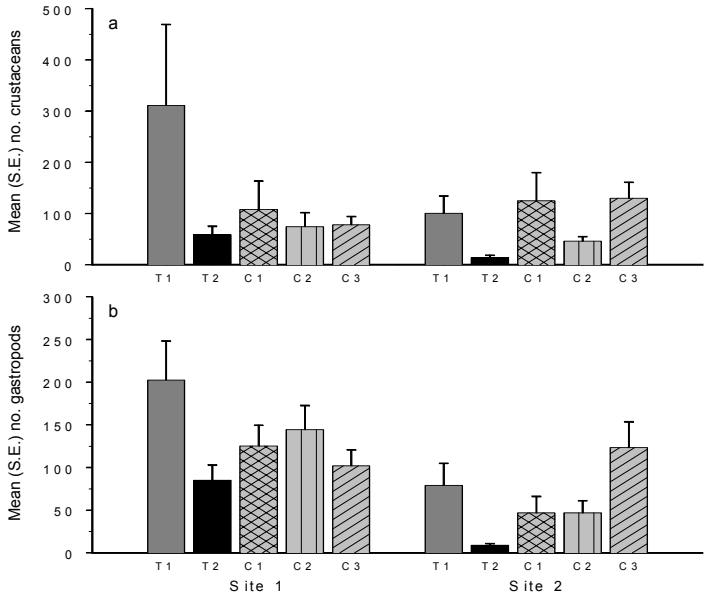

Figure 5. Mean numbers of (a) crustaceans and (b) gastropods in each of two Sites at Cape Banks; T1 no disturbance; T2 urchins removed; C1 urchins removed, partial cage; C2 urchins disturbed; C3 partial cage, urchins replaced; *n* = 4.

Finally, for the 10 taxa present in either (or both) of the two experimental treatments at Cape Banks and for the 13 taxa present in either (or both) of the experimental treatments at Jervis Bay, the number of taxa that were more abundant in the control treatment were calculated (Table 5). More taxa were more abundant in the control treatment at Cape Banks, although the opposite pattern was found in Jervis Bay (only at one site in Jervis Bay were these differences significant).

Table 5. The number of taxa more abundant in T1 (control) compared to T2 (full cage) and *vice versa* for each site in each location; n = 13 taxa at Jervis Bay and 10 taxa at Cape Banks.

		T1 > T2	T1 < T2	χ^2
Jervis Bay	Site 1	3	10	3.80
	Site 2	4	9	1.92
Cape Banks	Site 1	9	1	6.40**
	Site 2	7	3	1.60

Assemblage of gastropods

The gastropods, chitons and limpets were sorted to species at Cape Banks, where they were more numerous and diverse and where there appeared to be stronger patterns among treatments (Table 2).

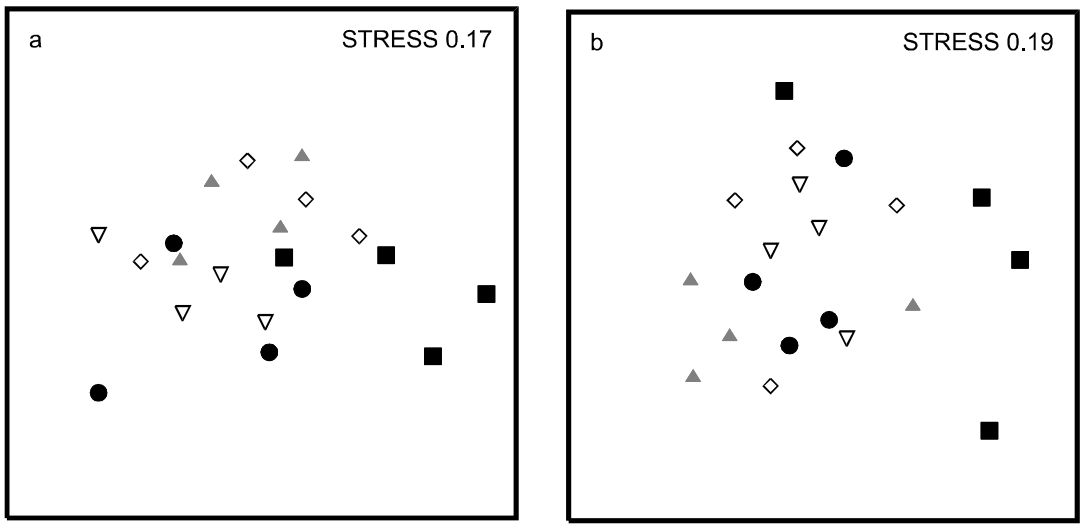

Figure 6. nMDS plots for all chitons, limpets and other gastropods in all treatments in (a) Site 1 and (b) Site 2 at Cape Banks; ● - T1 (undisturbed); ■ - T2 (full cage); ◇ - C1 (partial cage); ∇ - C2 (disturbance); △ - C3 (partial cage and disturbance); n = 4.

The data were analysed separately using npMANOVA on Bray-Curtis measures of dissimilarity from untransformed data (Treatments, fixed, 5 levels; Sites, random, 2 levels; n = 4). Although the interaction was not significant, the probability was very small (P = 0.06) and, in Site 1, the fully caged treatment was significantly different from each control treatment (P < 0.05 for each pairwise comparison), which did not differ among themselves significantly (P > 0.05). There were no consistent significant differences among treatments in Site 2. Nevertheless, in each site, three of the four fully caged treatments plotted separately from each of the controls, which were far more intermingled (Figure 6).

Therefore, differences in the assemblage between the control and caged treatments, when measured at coarse resolution were more strongly shown in Site 2 at Cape Banks,

whereas differences between these treatments, when chitons, limpets and other gastropods were identified to finer resolution (mainly species) was more strongly shown in Site 1.

The species that most contributed to dissimilarity between the two experimental treatments at Cape Banks (PRIMER; Clarke, 1993) were therefore selected for analyses of abundances.

Number of types of gastropods (including limpets)

The analyses were done for Cape Banks only, where all groups were analysed to fine resolution in all treatments in each site (Table 6). Similar resolution was used to compare only Treatments 1 and 2 (control and full cage) at Jervis Bay. In Site 2 at Jervis Bay, only the more widespread or abundant taxa were sorted (again in T1 and T2), but these data, (although internally consistent) cannot be directly compared to Site 1, nor with the data from Cape Banks. There were very few species of chitons, so they were not analysed separately.

There were no significant differences in the number of taxa of gastropods among Treatments (Table 6), although there were fewer taxa in the caged treatment in two sites at Cape Banks and one site at Jervis Bay (Fig. 7a, b).

Table 6. Analyses of the number of species of molluscs and abundances of chitons, *Clanculus brunneus*, *Clanculus cangulus* and *Eurytrochus strangei* per crevice in the two sites at Cape Banks; $n = 4$.

		Taxa		Chitons		*C. brunneus*	
Source of variation	**df**	**MS**	***F***	**MS**	***F***	**MS**	***F***
Site = S	1	697.2	11.68**	30.6	1.39	235.2	0.76
Treatment = T	4	87.2	1.01	31.5	1.59	210.2	2.57
S x T	4	86.3	1.45	3.4	0.16	81.7	0.27
Residual	30	59.7		22.0		307.5	

Chitons, test for Treatment after pooling Residual and S x T interaction.

		C. clanglus		*E. strangei*	
Source of variation	**df**	**MS**	***F***	**MS**	***F***
Site = S	1	1134.2	3.61	828.1	4.35*
Treatment = T	4	623.5	1.22	364.3	3.77
S x T	4	512.5	1.63	96.7	0.51
Residual	30	314.0		190.6	

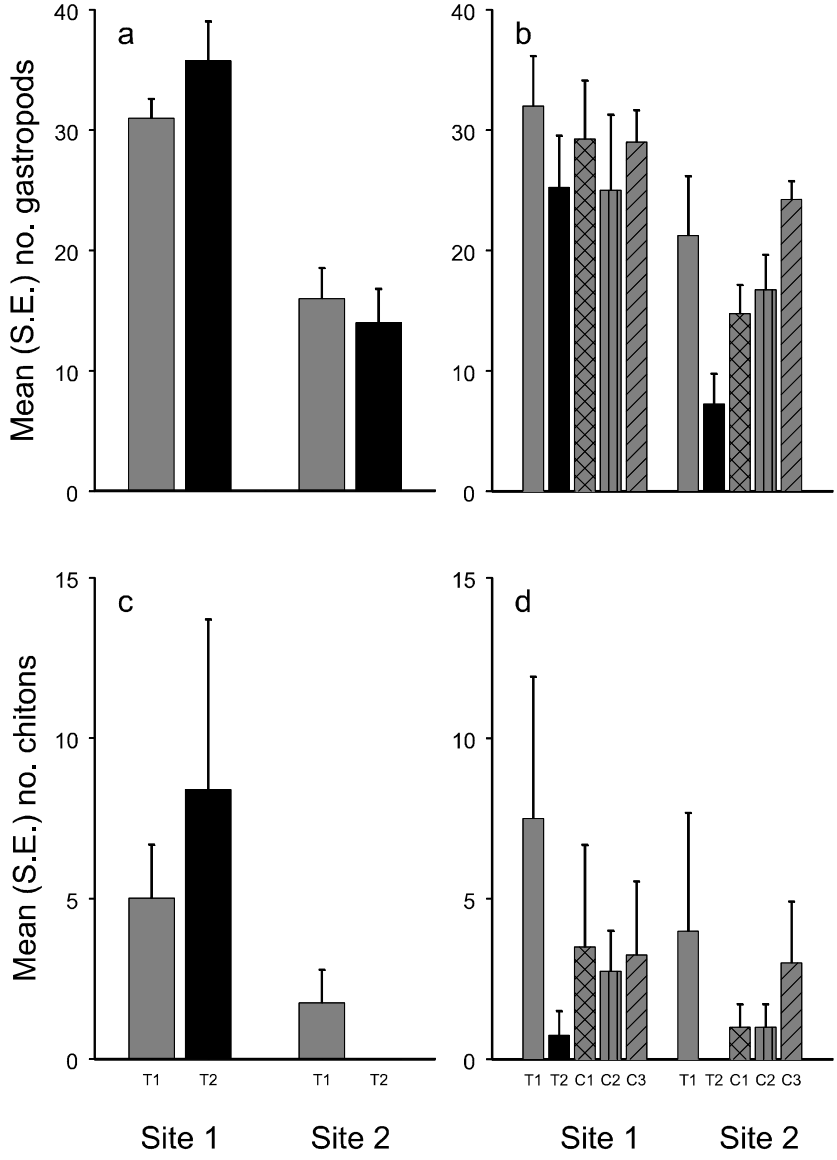

Figure 7. Mean numbers of (a, b) types of gastropods and (c, d) abundances of chitons (a, c) each of two treatments in two sites at Jervis Bay and (b, d) in each of five treatments in two sites at Cape Banks; T1 no disturbance; T2 urchins removed; C1 urchins removed, partial cage; C2 urchins disturbed; C3 partial cage, urchins replaced; *n* = 4.

Abundances of chitons and most important gastropods

The abundances of chitons and of the four gastropods identified in SIMPER as having contributed most to dissimilarities between T1 and T2 (together contributing 52 % and 47 % to Bray-Curtis dissimilarities for Site 1 and 2, respectively), were examined separately. One species, *Rissoina variegata*, was not analysed for both sites because it was only found in Site 2. For these measures of abundance, the data for each site in each location are comparable.

There were no significant differences among treatments for any analyses (Table 6). Nevertheless, the numbers of chitons, *Clanculus brunneus* and *Eurytrochus strangei* were smaller in the fully caged treatment in three of the four sites (Fig. 7c, d; Fig. 8a, b; Fig. 9a, b), *Clanculus*

clangulus was less abundant in the fully caged treatment in all sites (Fig. 8c, d) and *Rissoina variegata* was less abundant in the caged treatment in both sites in which it occurred (Fig. 9c, d).

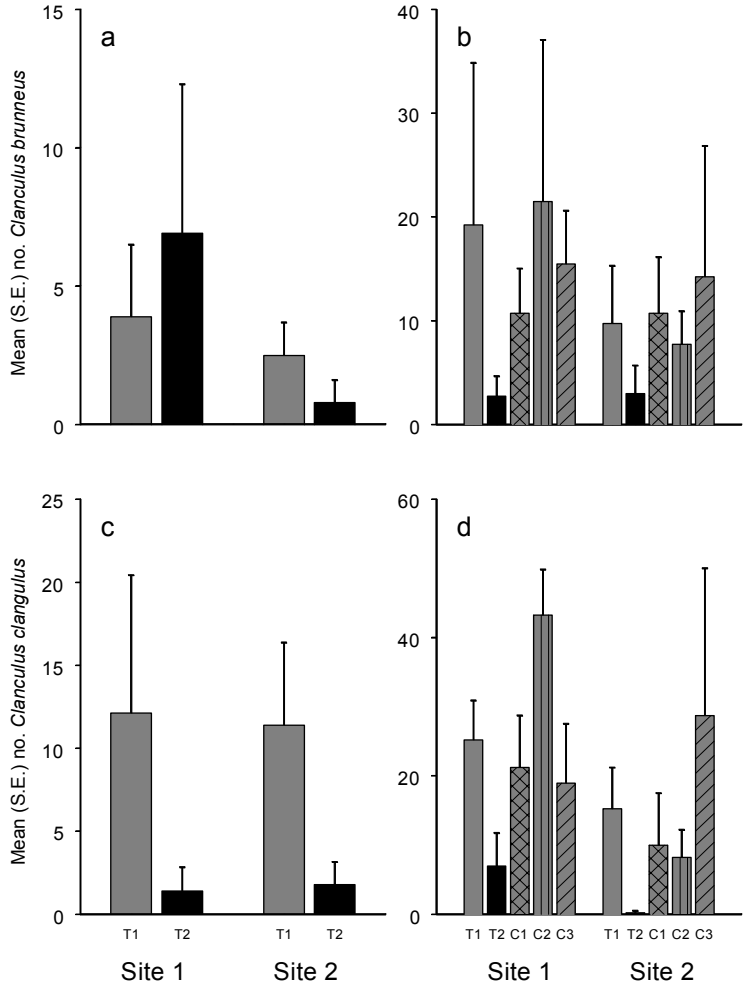

Figure 8. Mean numbers of (a, b) *Clanculus brunneus* and (c, d) *Clanculus clangulus* in (a, c) each of two treatments in two sites at Jervis Bay and (b, d) in each of five treatments in two sites at Cape Banks; T1 no disturbance; T2 urchins removed; C1 urchins removed, partial cage; C2 urchins disturbed; C3 partial cage, urchins replaced; *n* = 4.

Finally, for all species that numbered 10 or more in T1 and T2 in each site separately, the proportion that were more abundant in the control than in the fully caged treatment was calculated. Although only 50 % showed this pattern in Site 1 (of 54 species), 83 % of the species in Site 2 were more abundant in the control treatments.

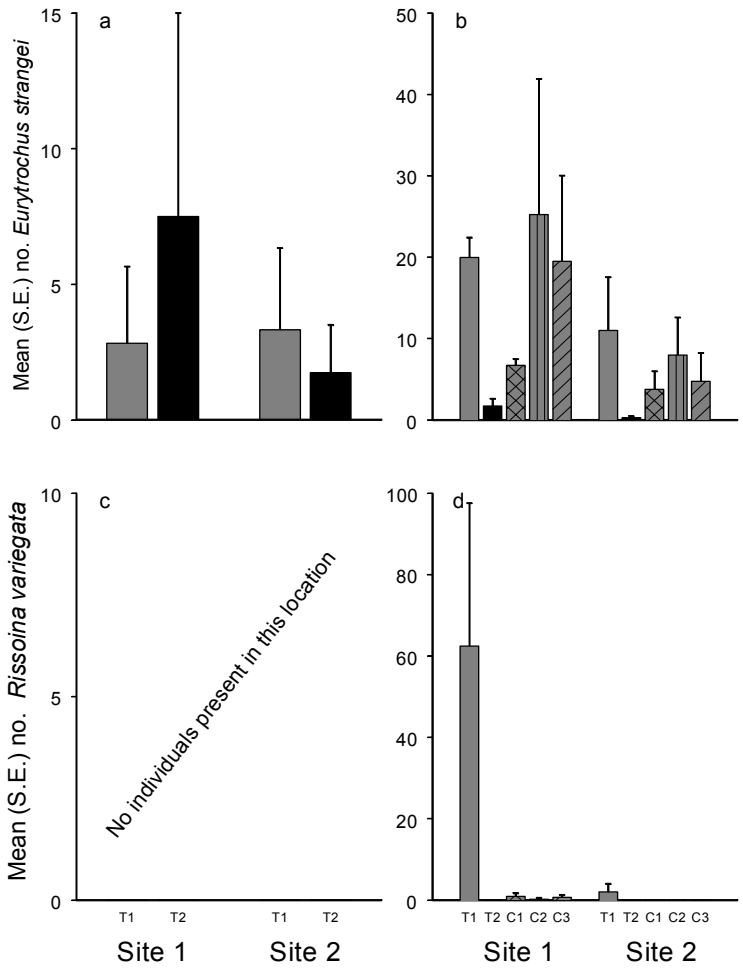

Figure 9. Mean numbers of (a, b) *Eurytrochus strangei* and (c, d) *Rissoina variegata* in (a, c) each of two treatments in two sites at Jervis Bay and (b, d) in each of five treatments in two sites at Cape Banks; t1 no disturbance; t2 urchins removed; c1 urchins removed, partial cage; c2 urchins disturbed; c3 partial cage, urchins replaced; *n* = 4.

DISCUSSION

The experimental procedures used here were appropriate to mimic fishing of sea urchins, even though the scale of the experiment was smaller than that of commercial fishing. Over a period of two months, smaller densities of urchins were maintained in the urchin-removal treatment than in the natural crevices. In addition, the cages used to exclude urchins did not have any other discernible effect on diversity (as shown by the similarity between the untouched crevices and the various procedural controls). Therefore, the cage itself, or the immediate disturbance of removing urchins, had no effect on types or numbers of taxa at the end of the experiment. All effects, therefore, can be attributed to there being small numbers of urchins in the caged crevices.

Although the results of these experiments were variable among the four sites and analyses were usually non-significant, there were clearly consistent effects of maintaining small densities of urchins in crevices on the associated faunal diversity. In both sites at Cape Banks, there were smaller numbers of gastropods and crustaceans (the two most diverse and abundant taxa). In addition, numbers of taxa of gastropods, plus abundances of chitons and the four most numerous gastropods, were smaller in the caged treatment than in the controls. This was shown in both sites. These produced differences between caged and uncaged treatments at the level of the faunal assemblage (in multivariate analyses), with significant differences among treatments found in Site 2, when coarse taxonomic resolution was used and in Site 1, when only gastropods were considered.

The patterns at Jervis Bay were not as consistent, but abundances of the chitons and selected gastropods in the caged treatment in Site 2 were smaller than in the uncaged control crevices. This pattern was, however, not found at Site 1. The causes of this spatial variability cannot be identified without further work, but the amount of sediment that accumulated in the crevices was extremely variable between Cape Banks and Jervis Bay. There was negligible sediment collected in the mesh bags at Cape Banks, so it was not quantified. In Site 2 at Jervis Bay, there was an average of 732 cm^2 of sediment per crevice; in Site 1, the mean amount was 2492 cm^2. Therefore, the effects of removing urchins may be stronger in areas where there is not a lot of sediment, although the generality of this cannot be determined without further experimental research.

Nevertheless, despite the spatial variability of these patterns, there are clearly strong effects of excluding urchins from crevices, in some places. These effects were generally to reduce diversity and/or abundances of particular taxa, particularly the more abundant taxa. This is a measurable effect of removing urchins on biodiversity. Therefore, a commercial fishery of *C. rodgersii* cannot be said to be "ecologically sustainable" until these effects are more fully understood and prevented. It is possible that these urchins offer protection from predators to small animals aggregating under the spines (Tegner and Dayton, 1977), or provide food to associated animals as they capture and eat drift *in situ* in the crevices (Lowry and Pearse, 1973). These (and other) alternative models, to be useful, must be able explain the strong effects of removal of urchins in some site and not in others. Until these models are further investigated, the effects of the fishery on biodiversity will not be understood.

The need to evaluate potential side-effects of cages or other disturbances associated with the experiment, reduced the number of crevices available for the two main experimental treatments. Therefore, replication was limited to $n = 4$ and data could only be analysed for the

most abundant and widespread taxa. It is extremely difficult to measure environmental impacts on sparsely distributed species because of the uncertainty associated with their absence from any site (review by Chapman, 1999). Further work to test hypotheses about the specific effects of removal of urchins might, therefore, reduce the number of procedural controls, in order the maximize the numbers of crevices in each treatment and, thus, to maximize measures of abundance for the rarer fauna. This is an extremely important point to consider further before commercial fishing be allowed/expanded because rare species are often considered more vulnerable to extinction (Gaston, 1994).

Therefore, if the fishery is to continue (i.e. not just if it is expanded), one or both of the following must be known:

(i) how to determine where there might be a problem, i.e. what makes some places show strong effects and not others?

(ii) how to support a fishery without causing these effects.

The sites are Jervis Bay differed from those at Cape Banks in a number of ways. Jervis Bay was more sheltered from waves, the crevices were not as deep (from the mouth to the base) and there was considerably more sediment, not only in the crevices (as described above), but also on the "barrens" surrounding the crevices. Any of these factors may explain the lack of a pattern at Jervis Bay compared to Cape Banks. For example, the urchins at Jervis Bay may spend more time actively foraging outside crevices (as a response to shelter from wave-action, or accumulation of sand in the crevices), which may limit the strong association between the urchins and other fauna. Alternatively, the sheltered site or the restricted depth of the crevices, may not provide adequate drift algae in the crevices, either to maintain the urchins or the associated fauna.

Until there is greater understanding of these issues, it is not possible to make decisions about where to allow fishing of urchins, nor where to set the limits to fishing in space (open *versus* closed areas) or in time (open *versus* closed seasons) and be certain of no adverse effects on biodiversity. The only way that the interaction between urchins and associated fauna can be understood is by further experimentation. For example, experiments that maintain different densities of urchins (rather than removal/control) will indicate what proportion of a population can be removed without adverse effects. Similarly, manipulation of depth of crevices to provide, for example, "Jervis Bay crevices" and "Cape Banks crevices" in the same sites, would provide data that might assist in selecting areas where fishing is less likely to affect diversity. Answers to these sorts of questions are needed by management before they can

provide adequate conservation, while managing the fishery. These answers will only be obtained via properly designed and replicated experiments, thus necessitating a strong collaborative interaction between the fisheries managers and ecologist (Peterson, 1993; Underwood, 1995).

These data also strongly suggest that there is an urgent need to measure biodiversity of the fauna that generally shares habitat with *C. rodgersii*, in areas that are currently being fished for urchins and those that are not. This is necessary to determine any larger-scale impact(s) than those suggested here.

The evidence from these experiments is that current fishing practices in the urchin fishery in New South Wales are not ecologically sustainable.

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