Interactions between the abalone fishery and sea urchins in New South Wales

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NON TECHNICAL SUMMARY

93/102 Interactions between the abalone fishery and sea urchins in New South Wales

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OBJECTIVES:

1. To quantify changes in the abundance of sea urchins and abalone on reefs in southern New South Wales
2. To describe the extent of the Barrens habitat and the degree to which it is expanding on reefs in southern new South Wales
3. To describe the influence of habitat type on the abundance of abalone
4. To describe the effects of habitat type, sea urchin density and harvest time on the quality of roe harvested from sea urchins and to gather preliminary data on the potential for a sustained commercial fishery for this species in New South Wales.

NON TECHNICAL SUMMARY:

A sampling method was developed to quantify changes in the abundance of abalone on reefs in NSW from Port Stephens to the Victorian border. The method is based on the repeated sampling of the same, fixed areas and has been used each year since 1994 to provide indices of the relative abundance of abalone of different sizes. The number and distribution of the fixed areas has been expanded each year since 1994. The relative abundance of small (<60 mm) and medium (<115 mm) abalone in NSW increased between 1994 and 1997. This information now provides an essential part of an annual stock assessment program for abalone in NSW, and is now funded by the NSW abalone industry.

A sampling program was developed to quantify changes in the abundance of sea urchins on reefs in NSW from Port Stephens to the Victorian border. The program is based on the use of random transects within Barrens habitat in a series of permanent sites to assess the abundance of sea urchins of different sizes. There was no evidence suggesting that the abundance of sea urchins (*Centrostephanus rodgersii*, hereafter *Centrostephanus*) increased in the four years of the present study. This duration of sampling is, however, unlikely to be sufficient to describe important changes in the abundance of sea urchins. For this reason, a comprehensive series of maps of the Barrens habitat were completed to provide a point-in-time

estimate of its areal extent.

The extent of the Barrens habitat on reefs in NSW differed along the coast, but overall it represented approximately 50% (SE = 3.9) of the nearshore reef between Port Stephens and the Victorian border. Barrens were most dominant in Disaster Bay where the represented 68% (SE = 6.7) of the nearshore reef, but just south of there only 1% (SE = 0.3) of the nearshore reef was Barrens. The time-scale of this study was not long enough to observe large-scale changes in the area of reef occupied by Barrens habitat, but a detailed baseline now exists for future comparison.

There was a strong influence of habitat on the abundance of abalone. Very few abalone were found in Barrens habitats, and abalone were most abundant in Fringe habitats. Manipulative experiments demonstrated that by reducing densities of *Centrostephanus*, Barrens habitat can be modified to alternative habitats that can enhance the recruitment, survival and growth of abalone. Such a strong negative interaction between an established high-value fishery, and an under-exploited resource with considerable potential for development, offers a rare opportunity to develop complimentary fishery management plans.

A variety of factors influenced the quality of *Centrostephanus* roe. The greatest yield of high quality roe was obtained from small urchins (60-80 mm) in the Fringe habitat during summer and autumn. Roe harvested from larger urchins, from the Barrens habitat, or during the spawning season, was generally of a poor quality. Because of the logistic difficulties and expense associated with processing sea urchin roe for markets, the fishery will need to consistently harvest roe of a high quality. Preliminary experiments suggested several techniques to enhance roe quality, in areas or times where it is poor, enabling the more reliable harvest of higher quality roe.

The development of the sea urchin roe fishery should be viewed within the larger context of their interaction with abalone, and the future of the abalone fishery. We recommend the further development of a management framework that facilitates an experimental approach to the management of both these fisheries. This approach should be carefully structured to allow both the development of a stock assessment program for sea urchins, and further research to help the development of the fishery. This would allow an increased sophistication in the management of both the sea urchin and abalone resources of NSW.

KEYWORDS: Abalone, *Haliotis rubra*, Sea Urchins, *Centrostephanus rodgersii*, Stock Assessment, Competition, Recruitment, Abundance, Habitat, New South Wales, Australia.

1. Introduction

1.1 Background

The New South Wales blacklip abalone (*Haliotis rubra*) fishery began as an open access fishery in the late 1950s and has followed a familiar development of increasing limitations on catch and effort since (Figure 1.1, reviewed by Andrew et al. 1996a). Presently, the fishery has a TACC of 333 t caught by 37 divers. The TACC has remained stable since 1995 and is now set annually by an independent committee, based largely on assessments by NSW Fisheries (Andrew et al. 1996a, 1996b, Worthington et al. 1997, 1998). Catch and effort in the fishery are reported on a fine spatial scale. Daily statistics are reported from 83 subzones in the state (Figure 1.2). For assessment purposes these are aggregated into 6 regions. The majority of catch is taken in the south of the state, from Batemans Bay to the Victorian border (Figure 1.3).

Related research on the fishery has concentrated on estimating growth (Worthington et al. 1995, Andrew et al 1997), optimal sampling of size-at-capture (Andrew and Chen 1996), the appropriateness of current size limits (Worthington and Andrew 1998a, b), and the analysis of catch and effort data (Worthington et al. 1998). As part of the NSW commercial abalone licence, divers are also permitted to harvest sea urchins and turban snails. The population biology of these latter species are relatively poorly described and importance of ecological interactions among species to the sustainability of the abalone fishery are insufficiently well-known to devise a comprehensive management plan for all three species.

Three species of sea urchin have been harvested commercially in NSW, *Centrostephanus rodgersii*, *Heliocidaris erythrogramma* and *H. tuberculata*. The purple sea urchin, *Centrostephanus* is the most common species in New South Wales and is a large, fast growing urchin that reach sizes up to 120 mm in test diameter (TD). *Heliocidaris erythrogramma* (the green sea urchin) is smaller, growing to 86 mm in diameter and although this species supports a commercial fishery in Tasmania, it is not commercially harvested in new South Wales. *H. erythrogramma* is found in a variety of habitats but is most abundant in the Fringe habitat in the immediate subtidal zone (Underwood et al. 1991). *H. tuberculata* (the red sea urchin) overlaps in distribution with *H. erythrogramma* along the NSW coast but is most abundant between Kiama and Ulladulla. There has not, until recently, been any commercial harvest of the red sea urchin.

The commercial harvest of sea urchins in New South Wales is small (Figure 1.4). Annual catches since the records began in 1989 were less than 1 t (green weight) until 1994 when nearly 5 t was harvested (Figure 1.4). In 1995 catches fell to approximately 1 t and remained at these levels. The majority of the harvest, since 1989, is thought to have been *Centrostephanus* although catches were not separated to species prior to 1995. In 1997, 66% of the total catch was *Centrostephanus* and the remainder *H. tuberculata.* Harvesting is restricted to 5 subzones in the abalone fishery. The development of markets for sea urchin from NSW has been slow, due largely to high labour costs in processing, low recovery rates and, to date, poor market acceptance. Alternative processing techniques and value-added products may be appropriate if exporting fresh product is not economic. Prior to the present study there were no

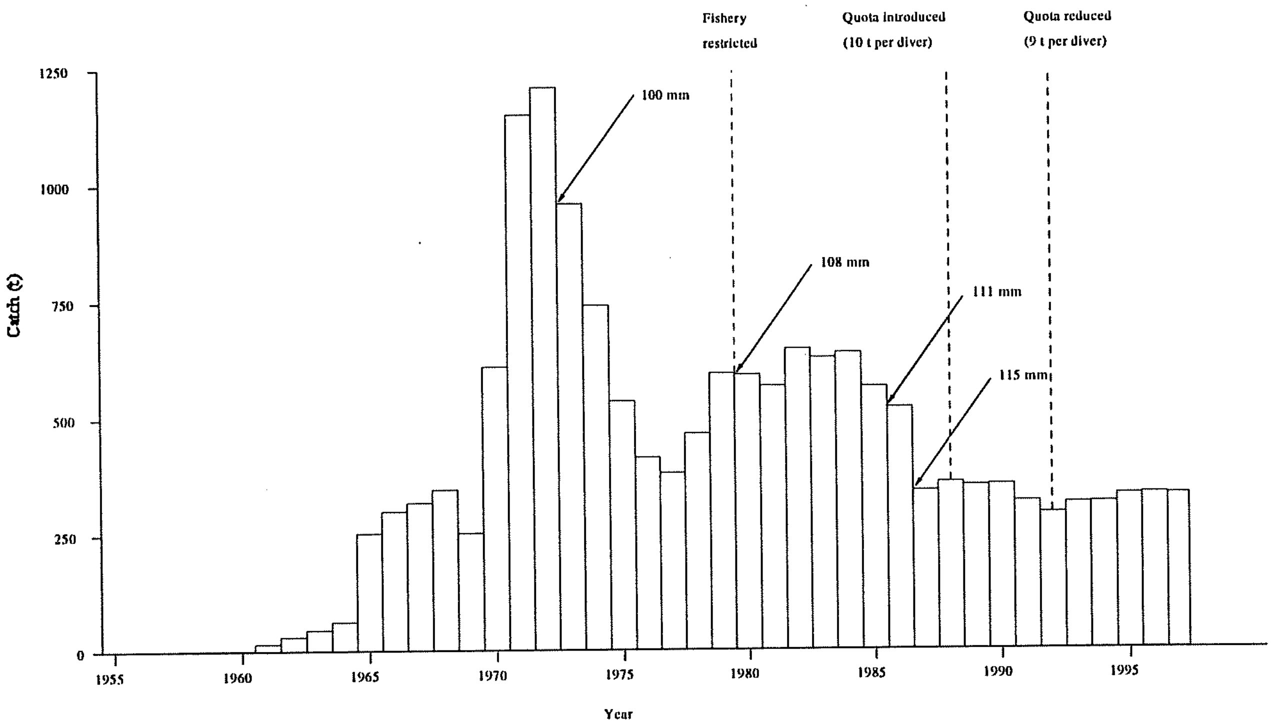


Figure 1.1 Total commercial catch and management changes in the NSW abalone fishery.

controls on the amount of urchins that could be harvested commercially nor where they could be harvested.

There are no estimates of the size or species composition of the recreational harvest of sea urchins. Daily catches are limited to 10 per person and are mainly collected from the lower intertidal zone and shallow subtidal. There is presently no minimum legal size enforced. Sea urchins may not be gathered in intertidal protected areas and other Marine and Estuarine Protected Areas.

1.2 Need

In New South Wales, *Centrostephanus* is the dominant herbivore on shallow rocky reefs. Its herbivory is responsible for the creation and maintenance of a major habitat, commonly termed Barrens or 'white rock areas' (Fletcher 1987, Andrew and Underwood 1989, 1993a, b, Andrew 1991, 1993, Underwood et al. 1991). Abalone are rare in this habitat (Shepherd 1973, Andrew and Underwood 1993, Underwood et al. 1991, McShane 1991, Andrew 1993). Commercial divers have claimed that sea urchins exclude abalone from the Barrens habitat and, more recently, that the area of reef covered by this habitat is increasing, at the expense of productive areas of reef. The great majority of available habitat in New South Wales contains few and scattered abalone. Most abalone are found in aggregations of varying size within crevices in the rock substratum or among boulders in the Fringe, *Phyllospora* forest or *Ecklonia* forest habitats.

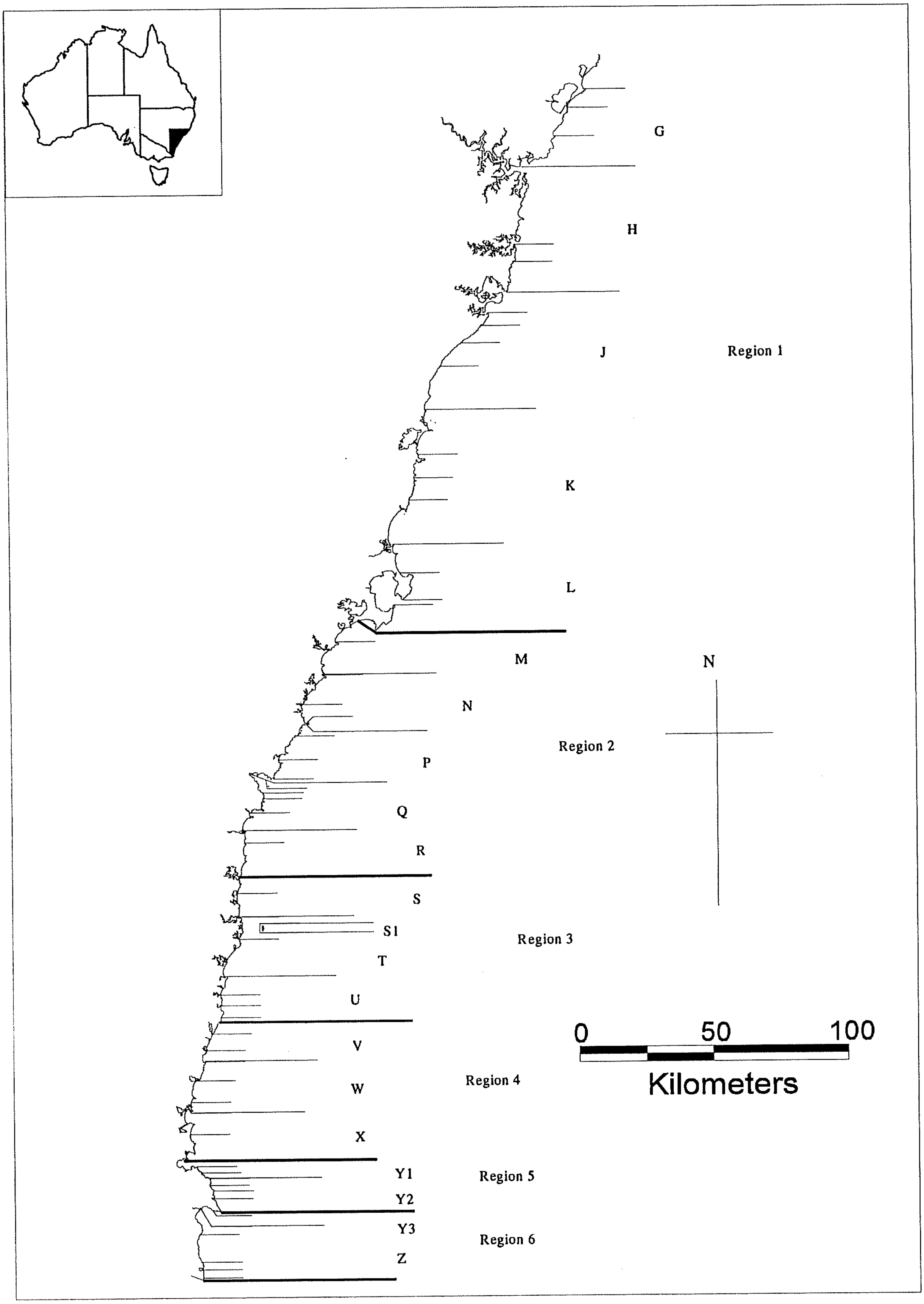


Figure 1.2 Map of the NSW coast south of Norah Head, showing catch subzones (small lines), zones (medium lines with letters) and stock assessment regions (large lines with numerals).

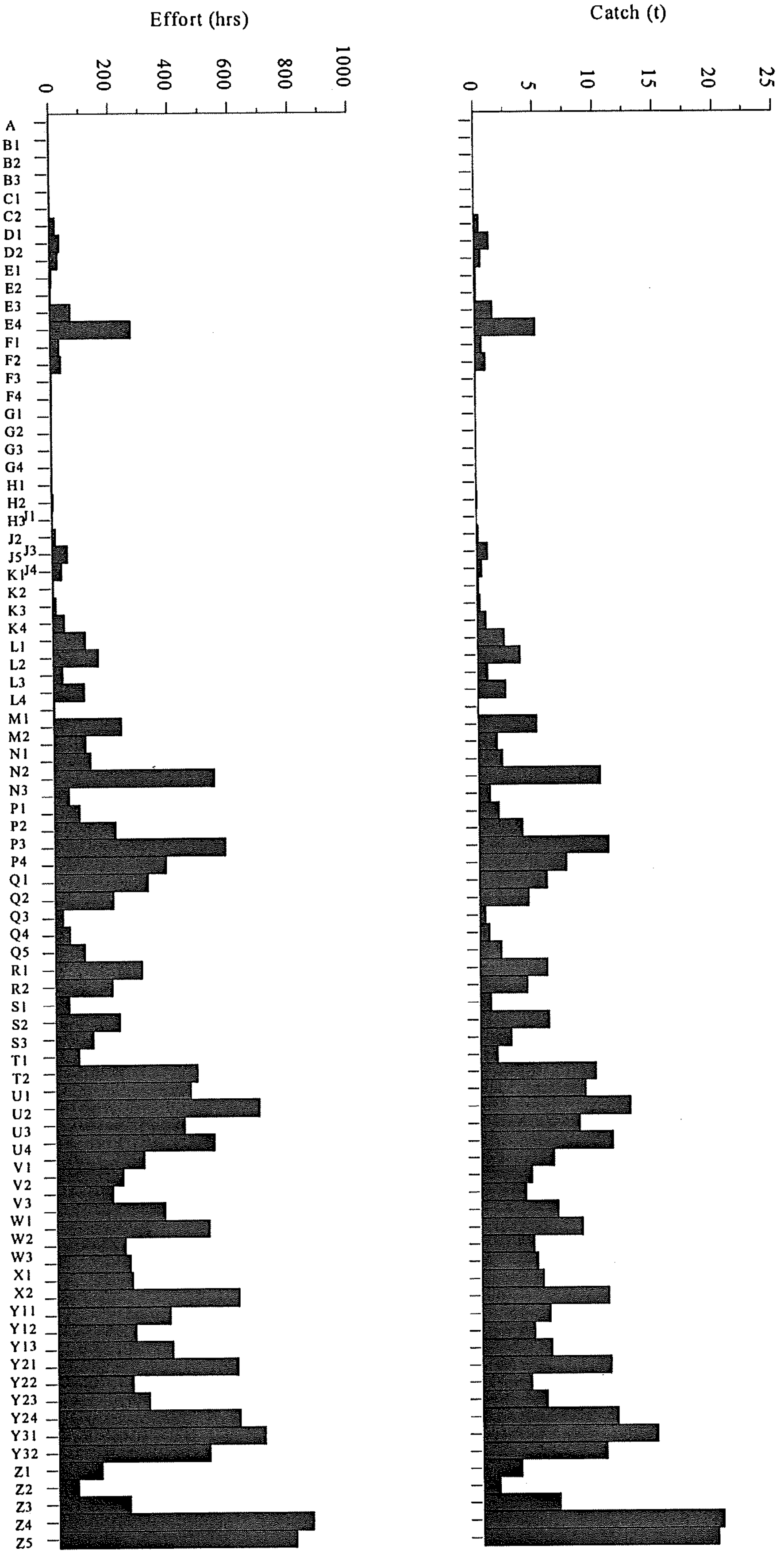


Figure 1.3 Patterns in A) total commercial catch and B) effort reported to subzone in 1997.

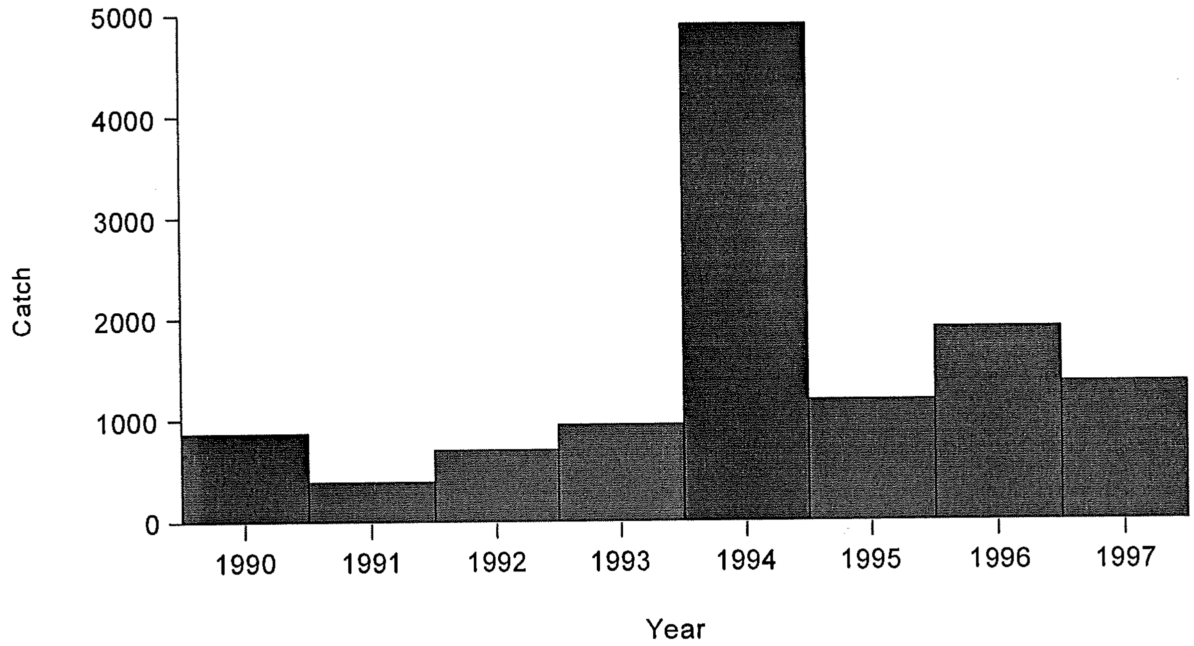


Figure 1.4 Total commercial catch (kg) of sea urchins in NSW between 1990 and 1997.

The strong negative association in the abundance of sea urchins and abalone, and the widespread occurrence of the Barrens habitat meant that a greater understanding of the relationship between these species was required for the effective management of both the established fishery for abalone and the developing fishery for sea urchins. There was a strong quantitative basis for research on the relationship between abalone and sea urchins on reefs in New South Wales (see above references) but little research had been done in the south of the state, where the abalone fishery is centred. Although the importance of interactions among fisheries is increasingly appreciated, rarely are the links between species so clear and their study so tractable as it was in the present case. The present research was designed to better quantify the extent of the Barrens habitat and better understand the processes underlying the observed negative relationship between abalone and urchins. This information was expected to facilitate management strategies to both increase the sustainability of the abalone fishery and allow the development of a compatible sea urchin fishery.

1.3 Objectives

1. To quantify changes in the abundance of sea urchins and abalone on reefs in southern New South Wales
2. To describe the extent of the Barrens habitat and the degree to which it is expanding on reefs in southern new South Wales
3. To describe the influence of habitat type on the abundance of abalone

4. To describe the effects of habitat type, sea urchin density and harvest time on the quality of roe harvested from sea urchins and to gather preliminary data on the potential for a sustained commercial fishery for this species in New South Wales.

2. Abundance of Abalone and Sea Urchins

Objective 1. To quantify changes in the abundance of sea urchins and abalone on reefs in southern New South Wales

A method to estimate the relative abundance of abalone was developed that repeatedly sampled fixed areas, referred to as crevices. This method allows annual samples of many crevices to be sampled across the geographic range of the fishery. These indices of density are considered representative of both the number of abalone in the crevices in that year and the relative abundance of abalone in the fishery. Using this interpretation, abundances of small abalone significantly increased between 1994 and 1996 in four of the six regions. There were also significant changes in the abundance of medium and large abalone, which differed among regions in the fishery. The greater precision of estimates of relative abundance made possible by this sampling design should lead to substantial improvements in both the power of tests to detect change, and analyses of the current status of stocks.

Methods to estimate the relative abundance of sea urchins are well-established. Changes in the density of *Centrostephanus* were estimated at sites at Eden and in Batemans Bay. Over the five years of sampling during the present study, there were no consistent trends in the abundance of sea urchins at these sites. Twenty seven sites between Port Stephens and Wonboyn estuary have been established to quantify long-term changes in the abundance and population structure of sea urchins.

2.1 Introduction

Methods to estimate the abundance of individuals, independent of the fishery, have become central to many modern stock assessments. Although estimates of absolute abundance are informative they are difficult to obtain and in the great majority of fisheries, patterns in relative abundance through time are used for stock assessment, along with additional information such as catch rates and other fishery-derived information (e.g. Andrew et al. 1996b, Worthington et al. 1997, 1998).

A range of fishery-independent methods have used to estimate the abundance of abalone, including timed swims (McShane, 1994, 1995), transects (Shepherd 1985, Andrew and Underwood 1992, Schiel et al. 1994, Gorfine et al. 1996) and mark-recapture techniques, including change-in-ratio methods (Kojima and Ishibasi 1985, Hirayama et al. 1989, Nash 1994). Several features of the ecology of abalone, not least their highly aggregated dispersion, ensure that the search for reliable methods will continue. Alternatives to area-based methods, such as timed-swims offer greater efficiency underwater (McShane 1994, McShane et al. 1996) but conversions to densities rely on a number of assumptions, particularly concerning the search efficiencies of divers. Formal comparisons of the performance of different methods are few (but see Hart and Gorfine 1997, Hart et al. 1997, Officer et al. in press). These authors concluded that area based methods provided the best compromise between repeatability and logistic efficiency, at least at the site studied in Victoria, Australia.

In New South Wales, abalone are most abundant in the shallow (0-10 m) subtidal zone of reefs, particularly in association with large foliose algae (e.g. kelps and turfing algae). Abalone are much less abundant in areas grazed by *Centrostephanus* (Andrew and Underwood 1989, Underwood et al. 1991, Chapter 3). Stratified, random sampling of these habitats is complicated by patchy nature of habitats in shallow water (Underwood et al. 1991, Chapter 3), and consequently, estimates of relative abundance at sites encompassing these habitats are often associated with high variances. A large part of this variation is caused by differences in the relative frequency of the habitats among replicate samples and therefore among years. An alternative approach, that repeatedly samples the same, fixed areas is likely to significantly reduce this source of variation, and hence lead to more precise estimates of relative abundance through time.

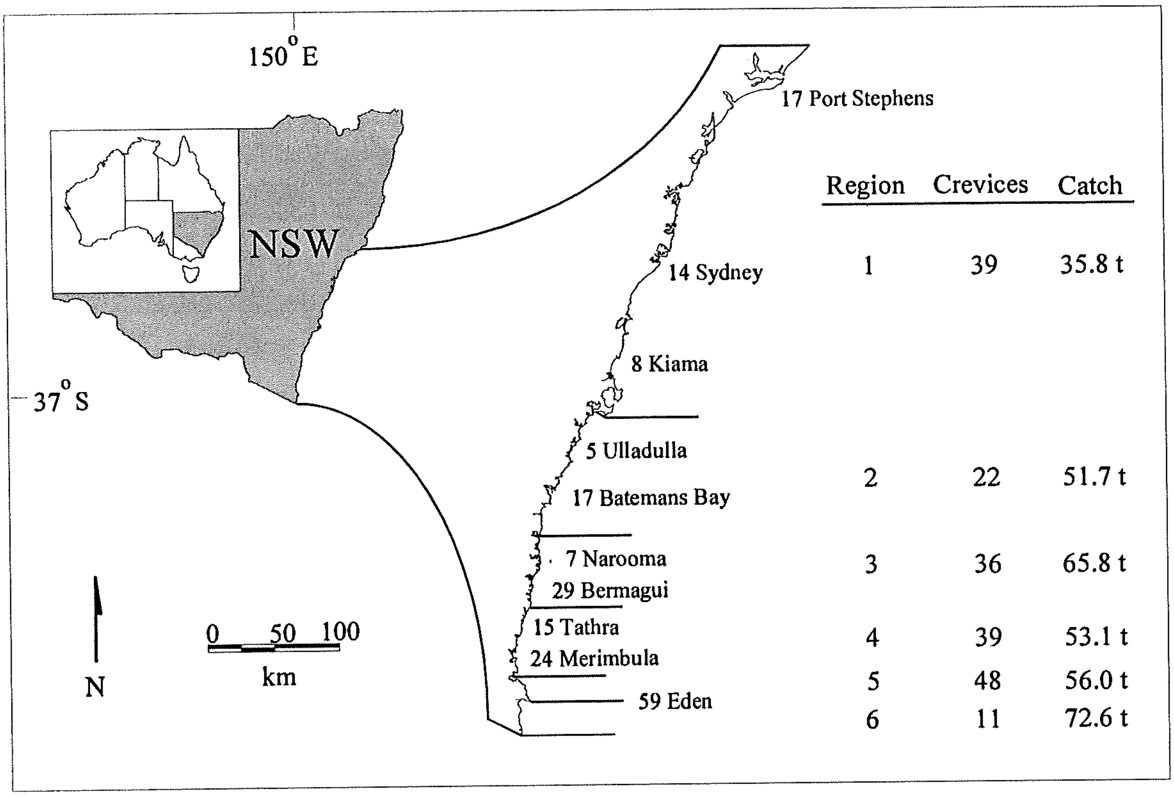
Here, we describe a method for estimating patterns in the relative abundance of abalone through time based on the repeated sampling of fixed areas. We present a generalised linear model to aggregate the estimates of relative abundance up to six regions within the fishery, and the whole fishery for three years. The design is also flexible enough to allow the incorporation of additional areas as they become available. Finally, we discuss potential problems with the sampling design, and suggest improvements that can be made to address them.

2.2 Materials and Methods

2.2.1 Position and size of crevices

A large proportion of the shallow, subtidal reef in NSW supports an appropriate habitat for abalone, but is inhabited by very low densities of individuals. Many abalone in NSW live in dense aggregations within small crevices and caves or under boulders. It is these aggregations that are mostly targeted by commercial divers and were chosen as the main sampling units. Sampling units (hereafter referred to as crevices) were chosen by searching shallow (i.e. <10 m), subtidal reefs between Port Stephens and the southern border of NSW. Each area of reef that could be mapped and clearly defined from surrounding habitat and containing more than 40 abalone was defined as a crevice. Effort searching for crevices was directed to areas with high commercial catch, so the distribution of crevices along the coast approximated the distribution of catch (see Figure 2.1).

The position of each crevice was carefully recorded to allow us to return to the same crevice each year. A detailed map of the extent of the crevice was drawn, together with details of the surrounding reef. Photographs of a diver on the surface above each crevice, with the adjacent rock platform in view, were also taken. Finally, the latitude and longitude of each crevice were determined by GPS and used to position the crevice on the appropriate map. These details proved adequate for the same diver to re-position each crevice at least one year after it was initially mapped, and, in many cases, allowed divers who had not visited the crevice before to independently sample them.



Region	Crevices	Catch
1	39	35.8 t
2	22	51.7 t
3	36	65.8 t
4	39	53.1 t
5	48	56.0 t
6	11	72.6 t

Figure 2.1 Map of Australia (inset) showing a section of the NSW coast and the distribution of crevices near major ports and within the six stock assessment regions in the fishery. The total catch in 1996 is also shown.

For each crevice, two replicate counts were made of all abalone in three length-classes. The length classes were small (< 60 mm), medium (60 -115 mm) and large (> 115 mm), and individuals were classified by reference to marks of an appropriate length on a slate. Where possible, the replicate counts were made by different divers (i.e. > 95% of counts). Most repeated counts were similar, but when counts differed by more than 10%, the diver with the lower count re-consulted the map and definition of the crevice, then re-counted. No rocks, boulders or abalone were moved during the counts, but drift algae was moved aside to reveal as many abalone as possible without disturbing the substrate. Oil-based crayons were used when necessary to prevent double counting or missing individual abalone.

These surveys were started in 1994 when 84 crevices were sampled and are now repeated annually. The number of crevices sampled has been increased and in 1997, 190 crevice were sampled. Many of these crevices occur in groups of 2-4 crevices separated by about 10-100 m, which we will refer to as clusters. At a larger spatial scale, the fishery is divided into six regions for stock assessment (Figure 1.2), and the crevices and clusters are distributed among these regions. As a result, the sampling design is able to describe changes in number of abalone at several scales. At the smallest scale, numbers of abalone can be described for each crevice in small, medium and large length classes, which can be pooled to total numbers of abalone. Changes in the number of abalone in any of these classes can then be scaled up to clusters, regions and the whole fishery.

2.2.2 Within and among year variation

As variation within a year has the potential to confound the comparisons of relative abundance among years, a subset of crevices were sampled repeatedly over two years to provide information on within-year variation. For 15 crevices in five clusters within Region 5, the number of abalone in the three length-classes was counted at 1-3 month intervals, between March and September in both 1995 and 1996. As there was nothing known about changes in abundance within a year prior to this study, to compare among years, crevices have been sampled between June and December each year. In addition, crevices were sampled at approximately the same time within this period each year.

To describe the number of abalone in each length-class, we searched for models in the generalised linear model framework (McCullagh and Nelder 1989). As is appropriate for such data, the number of abalone in each count were modelled as independent Poisson variables with a $\log_e$ link. An additional parameter was also included to account for any over-dispersion in the data. Several factors were forced into the model. These included a Crevice factor to account for differences in the number of abalone among the crevices, and Region and Year factors as these were of primary interest to the relative abundance of abalone. The model was then parameterised so that the number of abalone was described relative to that in the first year of the surveys, and *t*-tests were used to investigate the significance of parameters. The effect of a factor describing the divers who sampled each crevice was also investigated, but did not significantly reduce the deviance.

2.3 Results

2.3.1 Position and size of crevices

Crevices were distributed along the coast from Port Stephens to Wonboyn Beach (Figure 2.1). In Region 1, 39 crevices were spread over approximately 300 km, and clustered into 3 main areas adjacent to Port Stephens, Sydney and Kiama. In contrast, in the far south of NSW, 59 crevices were concentrated within the 50 km stretch of coast between Eden and Wonboyn Beach. Most of these crevices were located within Region 5, with only 11 in Region 6. The sampling design could be improved be devoting more effort to Region 6 and less to Region 5, in an attempt to distribute the crevices among regions in a similar pattern to total catch of the fishery. Despite careful searches, no appropriate crevices were found in the southern portion of Region 6, from Wonboyn Beach to the Victorian border. Estimating the relative abundance of abalone in these zones will require a different sampling method, probably a variant of the transect method used in Victoria.

The total number of abalone in each crevice ranged from approximately 40 to over 500 (Figure 2.2). Most crevices were inhabited by between 80 and 140 individuals. A large proportion of these individuals were in the medium length class, so that most crevices were inhabited by between 60 and 120 abalone between 65 and 115 mm long, although some crevices held more than 300 (Figure 2.2). Most crevices were inhabited by only 20-30 small individuals, although one crevice had over 130 (Figure 2.2). Large individuals, those larger than the MLS, were least common with most crevices supporting less than 15 individuals (Figure 2).

2.3.2 Variation within years

At the scale of individual crevices, changes in the total number of abalone within a year were significant and highly variable. For example, in two of the clusters, total numbers of abalone in each crevice often changed by greater than 10% among sampling periods separated by only two months, and in one crevice the total number of abalone changed by over 50% in one month (Figure 2.3). Despite these large and significant differences, patterns of change were not consistent among the crevices within each cluster. For example, the three crevices at South Leatherjacket showed very different patterns of change (Figure 2.3). Two of the crevices showed single, large changes, but at different times, and the third showed a gradual pattern of change through time.

At the scale of the cluster, changes in the number of abalone in each length class were also often significant and highly variable within a year. For example, there were significant changes of

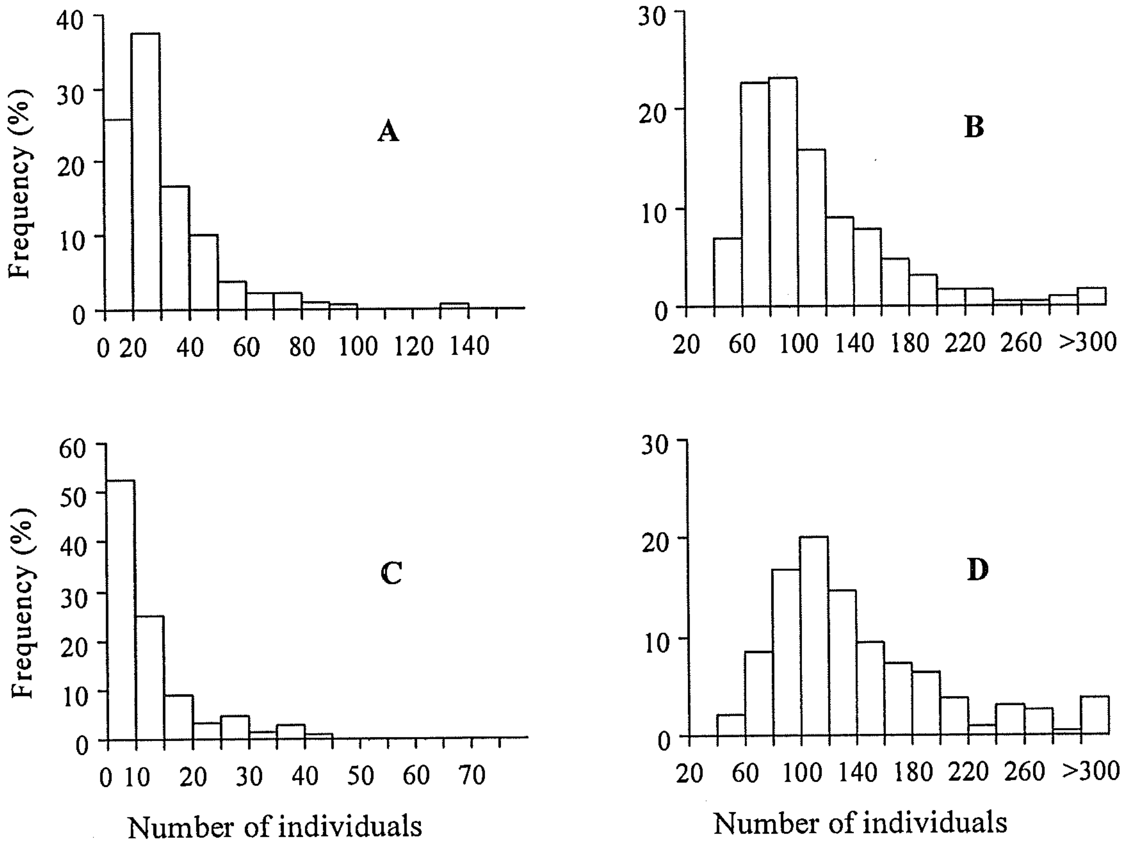


Figure 2.2 Frequency distribution of the mean number of abalone within all crevices from a) small length class, b) medium length class, c) large length class and d) total. There are 195 crevices in each distribution.

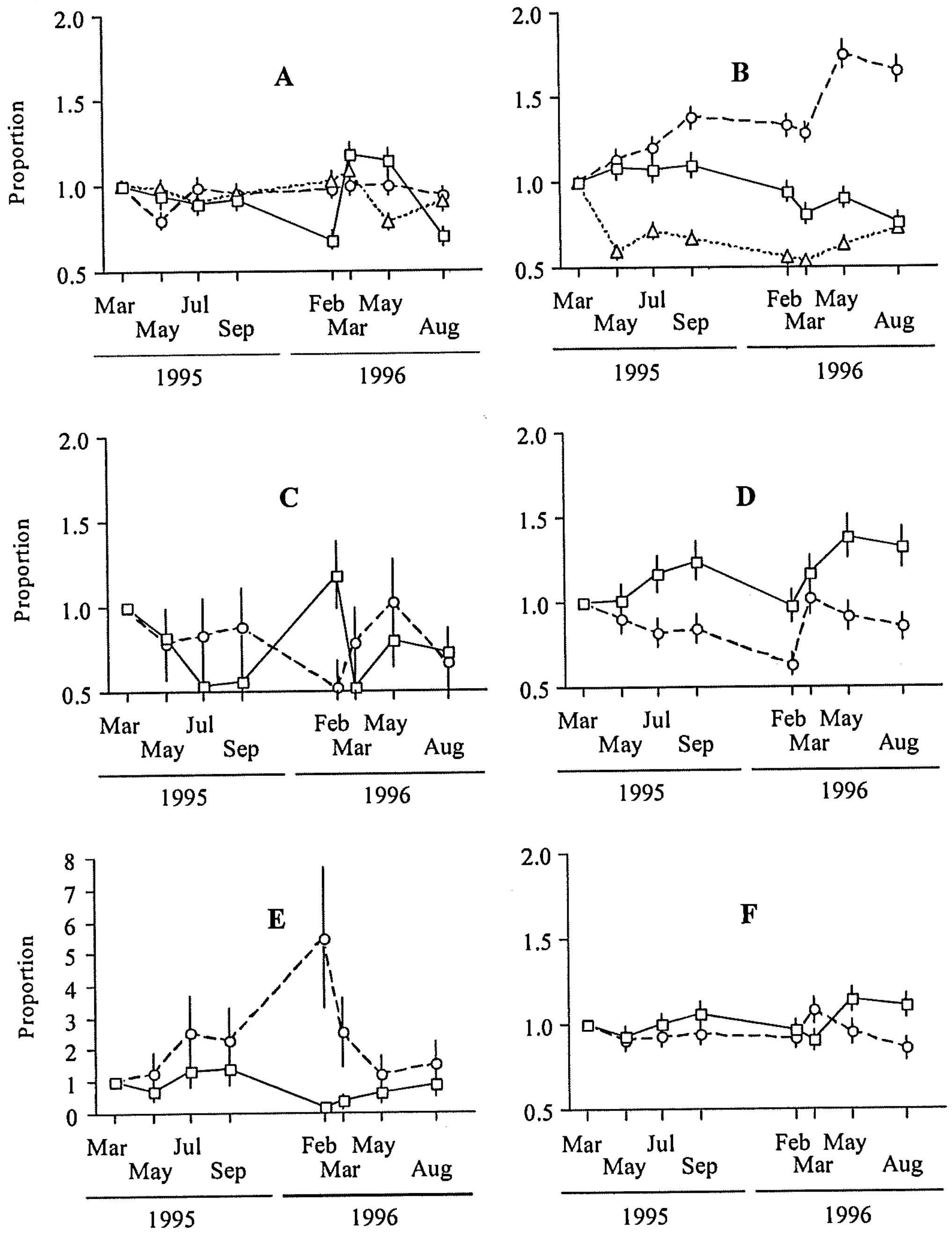


Figure 2.3 Relative number of a) total abalone within three crevices in the South Submarine cluster and b) the South Leatherjacket cluster, and the number of c) small, d) medium, e) large and f) total abalone within the South Leatherjacket (solid line) and South Submarine (dashed line) during eight sampling times. Bars are standard errors.

approximately 50% in the number of abalone in two months within small, medium and large length classes (Figure 2.3). Changes in the number of small and large abalone were particularly variable, partly because of the low number of individuals present in each crevice. This also resulted in, for example, a small increase in the absolute number of large abalone at South Submarine Rock being described as a >500% increase in the very low absolute number that were initially there (Figure 2.3). Change in the number of medium and total abalone within the two clusters were less variable, and patterns were more similar among the clusters (Figure 2.3).

At the scale of all clusters sampled, significant variation in the number of abalone was still evident, particularly for the small and large length classes (Figure 2.4). Changes of greater than 50% still occurred over periods of two months or less for both small and large individuals. There were no such changes in the number of medium abalone, but a >10% change did occur over a one month period (Figure 2.4). Total numbers of abalone showed no significant change among samples, or over the duration of the sampling period.

2.3.3 Variation among years

Since 1994 there have been changes in the number of abalone counted in most regions of the NSW abalone fishery. Numbers of small individuals increased between 1994 and 1995 in all regions, but this increase was significant only in Regions 1-4 (Figure 2.5). By 1996, the number of small abalone in each region was still above that in 1994 in all regions, but significant differences were only found in Regions 1, 3, 4 and 6. Numbers of medium abalone were not significantly different from those in 1994 in all regions and years, except for Region 2 in 1995 where numbers were significantly lower, and Region 4 (significantly lower) and Region 5 (significantly higher) in 1996 (Figure 2.5). Numbers of large abalone were not significantly different from those in 1994 in all regions and years, except for Region 3 and 6 in 1995, and Region 4 in 1996 where numbers were significantly higher (Figure 2.5). These differences in the number of abalone in the three length classes combined so that the total number of abalone in the crevices was not significantly different from that in 1994 in all regions and years, except for Regions 1 and 6 in 1995 and Regions 5 and 6 in 1996, where in each case, total numbers were significantly higher than in 1994 (Figure 2.5).

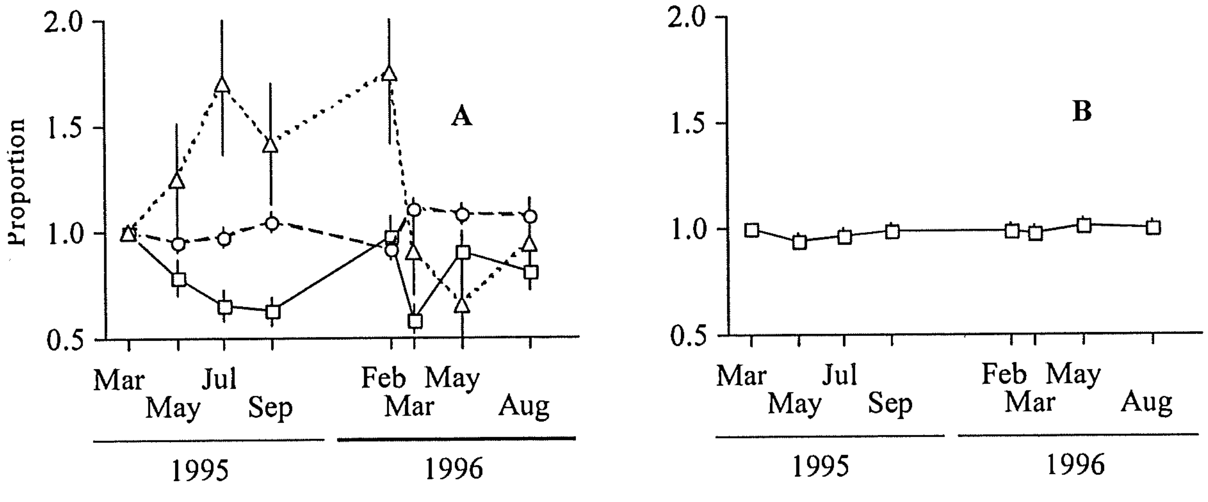


Figure 2.4 Relative number of a) small (solid line), medium (dashed line) and large (dotted line) abalone and b) total abalone within fifteen crevices during eight sampling times. Bars are standard errors.

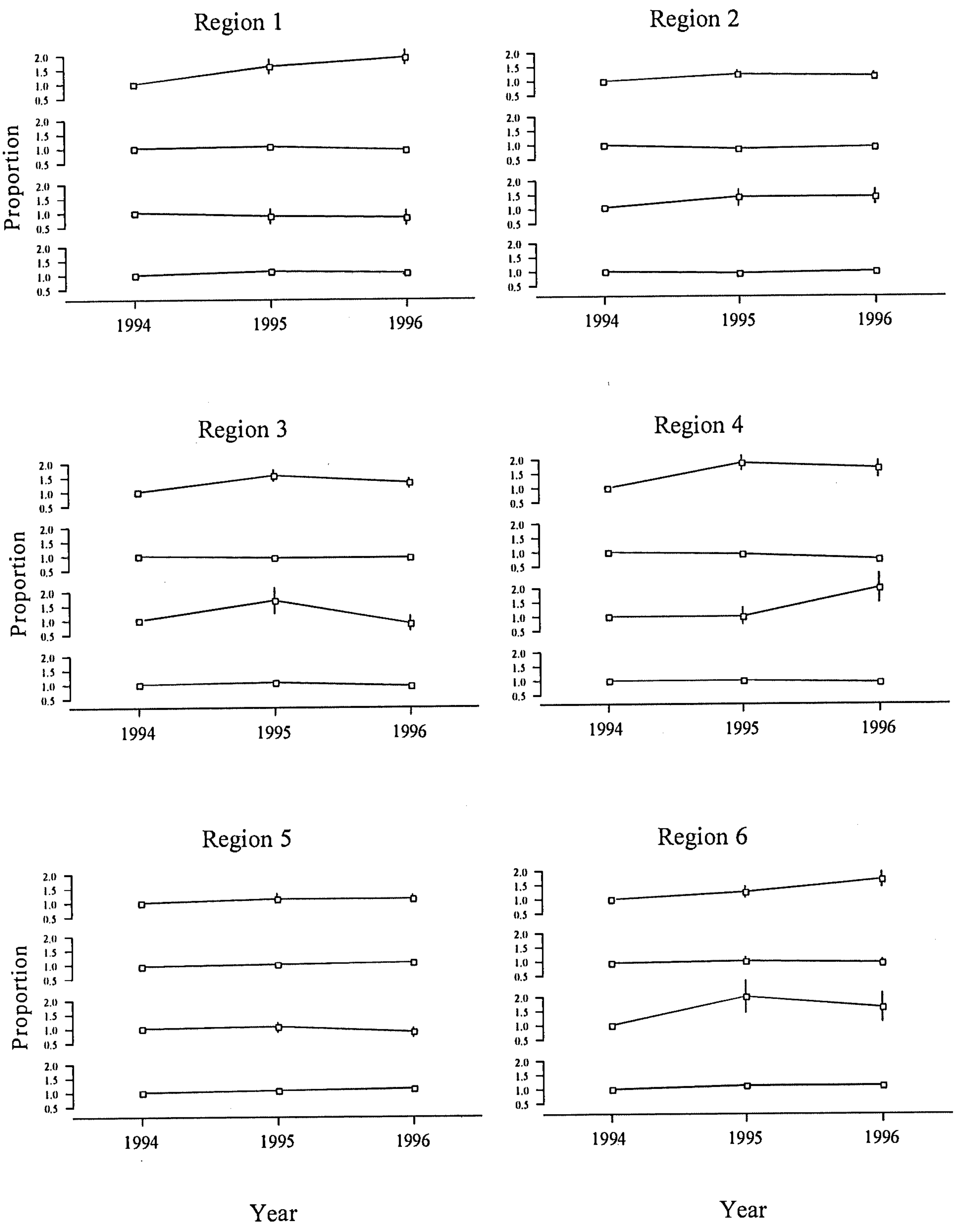


Figure 2.5 Relative number of small (top), medium (upper middle), large (lower middle) and total (bottom) abalone in crevices within each region. Sample sizes are shown in Figure 1, and bars are standard errors.

2.4 Discussion

2.4.1 Within and among year variation

There were significant changes in the number of abalone in three length classes within the crevices over periods of between one month and three years. At the smallest spatial scale, among individual crevices, changes in the number of abalone within years appeared unpredictable, and were not consistent among crevices within clusters. Among clusters of crevices, there was some evidence of consistent within year patterns, but apparently unpredictable change still occurred, particularly in the numbers of small and large individuals. At the largest scales, among regions and years, significant changes in the number of abalone within crevices were less common and often smaller. These changes represent the averaging of different patterns of change of the crevices and clusters within each region.

Within-year variation in the number of abalone in crevices has the potential to confound comparisons among years. For example, if consistent patterns of seasonal change occurred, and the crevices were not sampled at the same point in this seasonal pattern each year, differences among yearly samples would reflect both within and among year variation. Whilst we found large and significant changes in the number of abalone in both 1995 and 1996, much of this variation appeared unpredictable and not consistent among crevices, clusters or months. When this variation was averaged across crevices and clusters, there were comparatively few significant differences in the number of abalone. For example, there was no significant change in within year variation in the total number of abalone in the crevices. This suggests that by averaging patterns of change across crevices and clusters, and by sampling each crevice at a consistent time each year, comparisons among years should be valid. Such approaches are likely to be less successful for yearly comparisons of individual crevices because of the magnitude of within year variation at these spatial scales.

At the scale of regions, there were significant changes in the number of abalone in the crevices among the years. Numbers of small abalone were significantly higher than 1994 in four out of the six regions in both 1995 and 1996. In 1996 numbers of small abalone were more than 70% higher than those observed in 1994 in three of the six regions. This represents a substantial increase in the number of small individuals within these regions, suggesting strong settlement and survival of juvenile abalone. In addition to the regions where increases were significant, numbers of small abalone were greater in 1996 than 1994 in all regions, but in two of the six variation within the region made the differences not significant. Changes in the number of medium and large individuals were less consistent among the regions. There were some significant changes in the number of medium abalone. Significant decreases occurred in Region 2 in 1995 and Region 4 in 1996, whilst a significant increase occurred in Region 5 in 1996.

2.4.2 Interpreting change

The design of our sampling attempts to make inferences by sampling abalone at a random selection of sites and extrapolating to the abalone population within each region. This may be complicated by several factors. First, sites with abalone were not selected randomly, rather crevices were chosen as the sampling unit. Many abalone live outside crevices and cannot be sampled by this method. It is possible that patterns of change observed for abalone in crevices are not representative of abalone outside crevices. At some sites where random transects are possible, we have estimated patterns of change of abalone both in and out of crevices to allow comparison to those estimated from crevices alone. One possible factor that would allow the patterns of change of abalone in crevices to differ from the whole population is differences in the movement of individuals. For example, when the number of abalone in a crevice declines, due to fishing and natural mortality, individuals from the area surrounding the crevice may move in to the crevice. Consequently, the reduced abundance of abalone in the area will not be detected as the number of abalone in the crevice has remained the same.

Any net movement of individuals in to or out of crevices has the potential to effect our interpretation of patterns in the number of abalone in the crevices. Evidence from tagging experiments suggest movement rates of undisturbed blacklip abalone are very low (G. Hamer, unpublished data). In other words, a large proportion of abalone counted in a crevice in one year are likely to be counted again the following year. Regardless, it is still possible that some net movement may occur among years. Experiments removing individuals from crevices and monitoring patterns of change relative to unmanipulated crevices suggest some movement is possible into and out of crevices (N. Andrew et al. unpubl. data). Experiments monitoring tagged individuals, both within and adjacent to crevices, and their movement are planned.

Changes could be made to improve the representativeness of the sampling design. For example, because only those crevices with > 40 abalone were sampled, sparse, non-aggregated abalone were not sampled. These areas could be incorporated into the sampling design by setting up crevices with low numbers of abalone within them. In addition, to avoid the problems associated with small transects (i.e. high variances, Andrew and Underwood, 1989) areas outside of crevices could be sampled by long transects (e.g. 50 x 2 m) over the same fixed, but initially randomly chosen, stretches of coast. This would allow the sampling of abalone outside crevices, and also be able to better address questions related to the area of reef inhabited by abalone (see also Chapter 3).

By interpreting changes in the number of abalone in the crevices as reflecting changes in the abundance of abalone on the coast, several patterns in abundance can be described. Abundances of small abalone have increased significantly in four of the six regions between 1994 and 1996. This suggests there have been two strong years of settlement and early survival of juvenile abalone. It is also possible that these years of strong settlement will be reflected in an increased abundance of medium and large abalone in the coming years. Abundances of medium abalone have declined significantly in some regions, and increased significantly in others. These patterns may be reflecting past changes in the abundance of small abalone. As medium individuals are not exposed to legal fishing, changes in the number of medium abalone in crevices will reflect patterns in growth and natural mortality and illegal fishing. Illegal fishing appears to have occurred within a small proportion of the crevices, as indicated by large reductions (i.e. > 70%) in the abundance of large and medium individuals whilst most small individuals remain. Significant increases in the abundance of large abalone occurred in several regions. This suggests that the recruitment of individuals to the fishery has outweighed mortality due to natural causes and fishing. These changes in the abundance of large abalone have also been reflected in changes in the catch rates of the commercial divers. For example, catch rates of commercial divers have showed large and significant increases in Region 3 during 1995 when abundances of large abalone had also significantly increased (see also Figures 5 to 7, Worthington et al. (1997)).

2.5 Changes in the Abundance of *Centrostephanus rodgersii*

Methods to estimate the abundance of sea urchins are well-established (Andrew and Underwood 1989). Changes in the density of *Centrostephanus rodgersii* were estimated in the Barrens habitat at sites at Eden (inside North Head, inside South Head, Leatherjacket Beach) and at Jimmy's Island, Batemans Bay . At each site, sea urchins were counted in ten 10 x 1 m randomly placed transects. The transects were sampled by divers swimming slowly along the tape measure searching for sea urchins, but not overturning boulders or otherwise disturbing the reef structure. Sampling at Twofold Bay, Eden was done approximately bimonthly between November 1993 and July 1997. Densities at Leatherjacket Beach were estimated between April 1994 and February 1994 and Jimmy's Island was sampled between November 1993 and March 1997.

2.5.1 Results and Discussion

Over the 5 years of sampling during the present study, there were no consistent trends in the density sea urchins at either Eden or Batemans Bay (Figure 2.6).

This result is consistent with that described by Andrew and Underwood (1989) for exposed sites at Port Stephens, Sydney and Wollongong. The five years sampled during this study may be too short to adequately quantify changes in density caused by variability in mortality and recruitment. For this reason, 27 sites between Port Stephens and Wonboyn estuary have been established to quantify long-term changes in the abundance and population structure of sea urchins. These sites, and estimates of density and population structure are described in detail in Chapter 3.

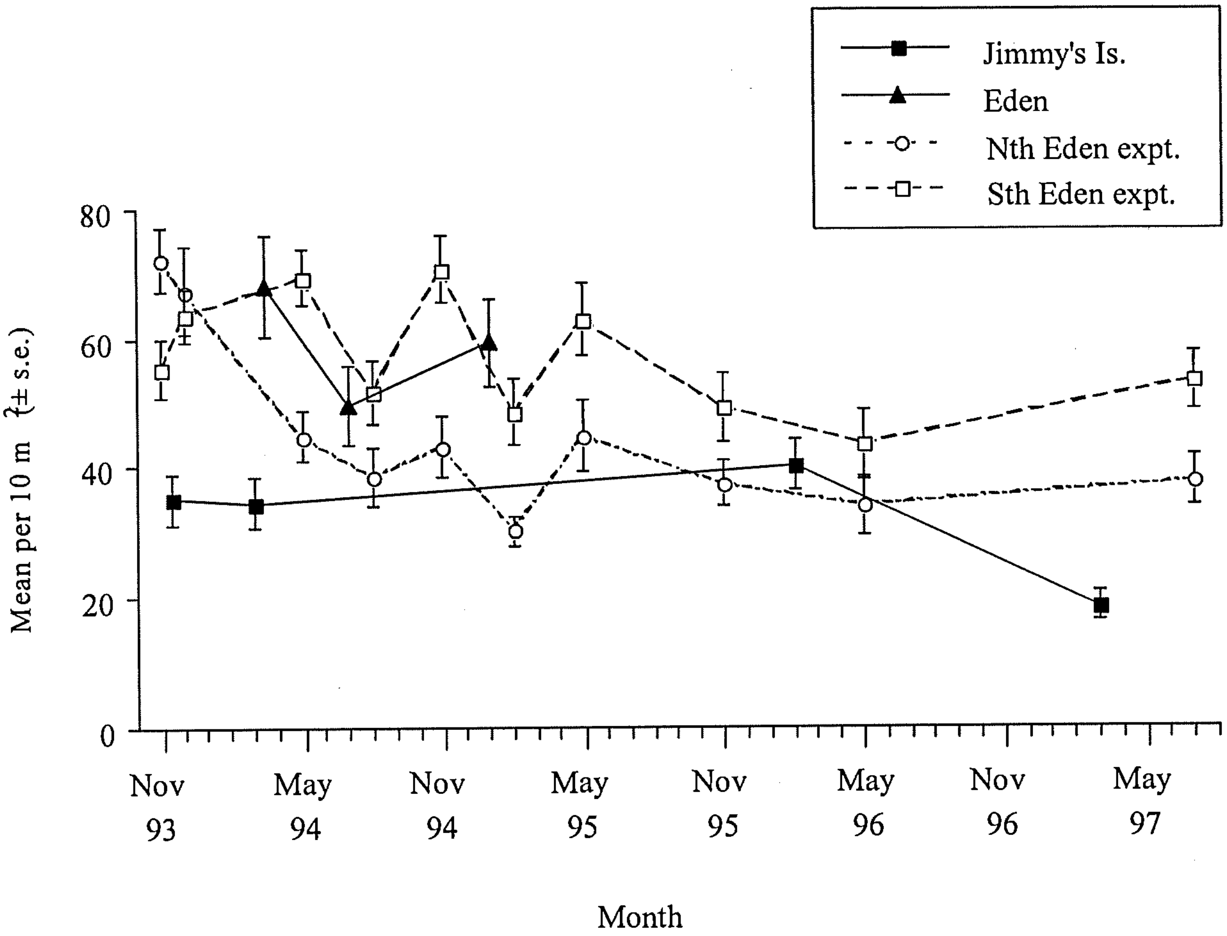


Figure 2.6 Mean number (± s.e.) of urchins per 10 m in control sites for sea urchin removals at Jimmy's Is., Eden and the experiment at the north and south sides of Twofold Bay (Eden).

3. The Extent of the Barrens Habitat and Habitat Representation

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Objective 2. To describe the extent of the Barrens habitat and the degree to which it is expanding on reefs in southern New South Wales.

In 1996-7, the Barrens habitat covered an estimated 50% (SE = 3.9) of nearshore reefs between Port Stephens and Wonboyn Beach, New South Wales. Coverage was greatest in Disaster Bay where 68% (SE = 6.7%) of nearshore reef was Barrens Habitat. South Of Wonboyn Beach, only 1% (SE = 0.3%) of the nearshore reefs were Barrens. The reefs mapped at Cape Howe, Nadgee, and at Turingal were significantly smaller than those at Port Stephens, Sydney, Bass Point/Kiama, Jervis Bay, Ulladulla, Batemans Bay, Haycock Peninsula, Eden, and Disaster Bay.

Comparison with existing transect-based estimates of representation at Haycock peninsula suggest there has not been an increase in the Barrens habitat at that location between 1988 and 1996-7. The coverage of previous estimates was not adequate to provide a more general comparison. To provide a baseline for future descriptions, 60 sites spread between Port Stephens and Victorian border were mapped to allow quantification of habitat change over the long-term. The densities and size-structure of *Centrostephanus* at 27 of these sites have also been estimated to quantify the relationship between the demography of this species and the area of the Barrens habitat.

3.1 Introduction

Based on earlier descriptions (Larkum 1973, Shepherd 1973, Farrant and King 1982), Underwood et al. (1991) described seven habitat types found on shallow subtidal reefs along the coast of New South Wales. These habitats were based on patterns in the association and relative abundance of a relatively small number of species. Prominent among these were several species of large brown algae and sea urchin. These species either provided the dominant character of a habitat (e.g. large brown algae) or were recognised as important agents of change (e.g. sea urchins). There were depth and latitudinal trends in the occurrence and representation of these habitats along the NSW coast (Underwood et al. 1991). This categorisation of habitats has proved to be a useful framework for studying the ecology of subtidal rocky reefs in NSW.

The ecology of subtidal rocky reefs in NSW is strongly influenced by several species - a sea urchin (*Centrostephanus rodgersii*) and two species of large brown algae (*Ecklonia radiata* and *Phyllospora comosa*). Much is known about the ecology of the former two species and almost nothing about the ecology of the latter. The contribution of sea urchins to the maintenance of habitat structure has been documented in a series of studies (Shepherd 1973, Fletcher 1987, Andrew 1991, 1993, 1994, Andrew and

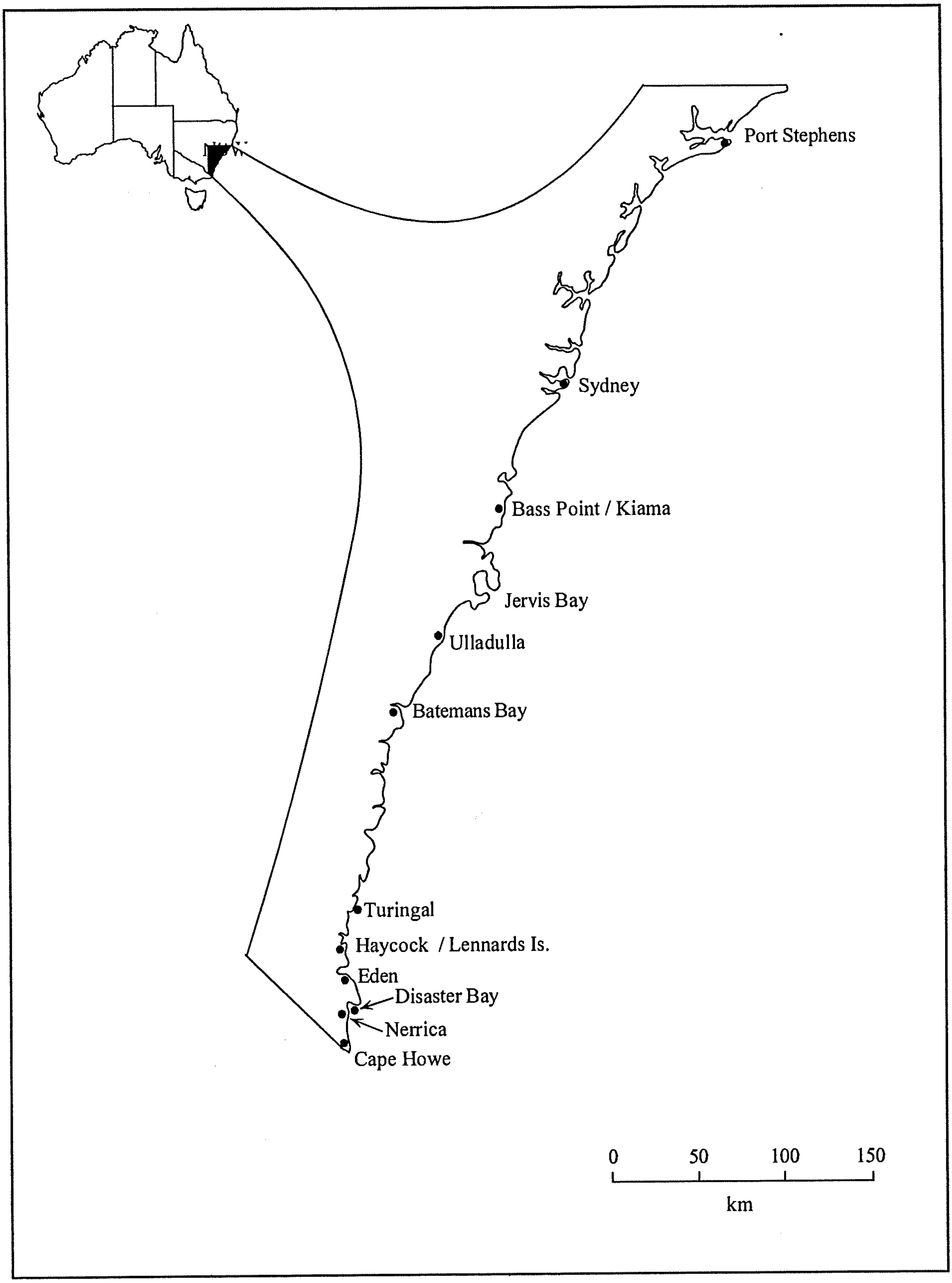


Figure 3.1. Location of mapped sites along the New South Wales coast.

Underwood 1989, 1992, 1993, Jones and Andrew 1990). This habitat type shares features in common with those in other parts of the world, notably the absence of large brown algae, high though variable densities of sea urchins and assemblages of fish very different from those in neighbouring areas of reef with a large biomass of algae (op cit and Holbrook et al. 1991). Abalone are rare in this habitat (Shepherd 1973, Andrew and Underwood 1989, 1993b, Underwood et al. 1991, Andrew 1993). The areal extent and persistence of patches of Barrens has potentially large implications for the ecology of many species of algae, fish and commercially important invertebrates such as abalone.

The objective of this chapter was to describe the extent of the Barrens habitat in southern New South Wales. As outlined in the milestone reports, the method used to quantify the extent of the Barrens habitat was revised. Transects were replaced with aerial photographs which provided more representative estimates of the coverage of the Barrens habitat. The maps produced from these images provide a baseline for future examinations of persistence of patches of different habitat types on reefs in southern New South Wales. The maps have the added advantage of allowing changes in the location of boundaries between habitats to be quantified and facilitate testing hypotheses about the impact of spatial structure on community processes. Of particular interest was the Barrens habitat because of the clear negative association between *Centrostephanus* and abalone. For this reason a subset of the sites mapped was further sampled as the basis for a long-term study of the links between the demography of *Centrostephanus* and the persistence of the Barrens habitat.

3.2 Methods

3.2.1 Locations and Sites

Twelve locations were selected between Port Stephens and the Victorian border (Figure 3.1). The locations were spread along the coast to achieve a wide geographic spread and were separated by an average straight line distance of 46.5 km (SD = 49.4). Locations were concentrated in the south of the state where the abalone fishery is largest. Selection thereafter was limited by the presence of rocky reefs (an estimated 33% of the NSW coastline is rocky reef, Fairweather 1990) and availability of adequate aerial photographs. Sea conditions and the topography of the reefs meant that some stretches of coast were excluded. For example, the reefs off the front of Jervis Bay were in water too deep to be photographed adequately and, in the far south of the state, poor photograph quality excluded stretches of coast between Bermagui and Tathra, and between Mowarry Rock and Green Cape.

Five sites, each between 4 and 5 hectares in area were mapped per location. Within the limits provided by the availability of adequate photographs, sites were selected haphazardly by applying the following criteria: (1) Each site was separated by a minimum of 200 m along the coast or around headlands, (2) sites were no closer then 100 m from a beach, and (3) offshore islands, harbours and estuaries, and other sheltered waters were excluded. The average straight-line distance between sites within a location was 2.4 km (SD = 3.6). The reef at each site was mapped to 150 m seaward from the nearest point in the intertidal zone visible at the time the photograph was taken (hereafter defined as nearshore reef). The

150 m boundary was determined on an *ad hoc* basis after inspection of the full set of aerial photographs indicated that features of the reef could be seen at least 150 m shore in all images. Maps at all sites were therefore limited to 150 m from shore to avoid biases introduced by differences in visibility. Such biases were possible because of the depth related distribution of some habitats in New South Wales (Underwood et al. 1991). Contiguous reef extended beyond this 150 m boundary in 40% of the 60 sites mapped. These limitations in sampling mean that several habitats were under-represented at those sites where the reef extended beyond 150 m. It is most likely that the *Ecklonia* forest and the Deep Reef habitats were most affected and the latter habitat was not recorded in this study.

The aerial photographs were taken within two hours of low tide on days with low swell and relatively calm sea surface conditions at times of the year or day with low sun angle (to avoid reflections off the sea). These constraints severely limited the number of appropriate days and this was compounded by the long length of coastline which meant that it was impossible to satisfy these conditions for the whole coast on a single day. The photographs were completed in a series of runs in 1996 (on 12/7, 6/8, 9/8, 18/9, and 15/10) and 1997 (29/4, 4/6, 9/6, and 10/10). The photographs were taken by BHP Land Technologies Ltd using a Wild RC10 camera and 214 mm lens. All photographs were taken at 1: 8,000 except those in Sydney which were taken at 1: 8,500 because of flight restrictions in the vicinity of Sydney International Airport.

Contact prints were digitised at 400 dpi to produce 24-bit colour TIFF files. The contrast in the resultant images was improved using a Guassian enhancement in DIMPLE image processing software (Process Software Solutions Pty Ltd). The images were geo-rectified to the Australian Map Grid (AMG) using ground control points obtained by differential GPS. A minimum of three ground control points were obtained per image. Registration defined AMG points for each pixel in the image and resampling produced images with all pixels with an area of approximately 0.5 m^2. This procedure was accurate to within one metre.

Mapping was done in ArcView (Environmental Systems Research Institute Inc.) by tracing the boundaries of visibly different patches of habitat. A mixture of enhancements and magnifications were used. This was accomplished in several iterations, the first tracings were done purely on the basis of colour by someone unfamiliar with the locations or the ecology of these reefs. These preliminary maps were taken into the field and 'ground-truthed' by divers. The GIS files were then revised in accordance with the corrected maps in consultation with those divers and people with detailed knowledge of the reefs.

The ground-truthing could not assess the accuracy of the location of boundaries between patches of different habitat types without differential GPS coordinates. This was beyond the logistic capacity of the present study. The location of boundaries was therefore assumed to be correct and the ground-truthing was used to assign habitat categories to the patches. In a few instances, patches of reef that appeared the same in photographs were, in fact, comprised of different habitats. Subdivision of these was done as accurately as possible in the field based on the preliminary maps. This process worked and was largely based on physical features of the reef and the shapes and location of surrounding habitat patches and the intertidal zone.

The maps were ground-truthed as soon as possible after the creation of the preliminary maps but sometimes up to 11 months after the photograph was taken. The high degree of congruence between the preliminary maps and the patterns observed during the ground truthing suggests that there was little change in habitat structure during the intervening period. One exception to this was found at a site at Nadgee where there was a large sand inundation in the 23 weeks between the photograph being taken and when the site was ground-truthed (see below).

Seven habitat categories were recognised by Underwood et al. (1991): *Pyura*, Fringe, *Phyllospora* forest, Barrens, *Ecklonia* forest, Turf and Deep Reef. These categories were used in the present study and augmented with an additional habitat type, *Durvillaea.* This addition was required because the bull kelp *Durvillaea potatorum* formed dense bands in the low intertidal and immediate subtidal zone of many reefs in the far south of the state. The Turf habitat, as defined by Underwood et al. (1991), was largely defined by a high percentage cover of articulated coralline algae (typically *Corallina* spp. and *Amphiroa* spp.) and the presence of a wide variety of large brown algae such as *Ecklonia radiata, Sargassum* spp., and *Cystophora monoliformis*. These latter species are at lower densities and this habitat is differentiated from other habitats dominated by large brown algae by the absence of a canopy. The Turf habitat, as defined in Underwood et al. (1991) and used here is therefore more diverse than the other habitat types, being loosely defined by the absence of a dominant species.

A subset of the 60 sites mapped as part of this study was further sampled to provide the basis of a long-term study of the processes underlying the persistence of the Barrens habitat. The abundance and size structure of populations of *Centrostephanus* were quantified in patches of Barrens habitat at three sites at each of nine locations (Disaster Bay, Eden, Haycock Peninsula, Batemans Bay, Ulladulla, Bass Point, Sydney, and Port Stephens). Turingal, Nadgee and Cape Howe were excluded because of the absence or rarity of Barrens habitat at the sites surveyed. Three sites within each location were randomly selected from the five sites mapped. At each site, sea urchins were counted in 15 (10 x 1 m) transects. Fifty metre tape measures were haphazardly laid out and start positions for the transects randomly chosen along either side of the transects. In the same area as the transects were done, the test diameters (TD) of a minimum of 150 *Centrostephanus* were measured. Sea urchins were found by careful searching without disturbing the substratum.

Differences in the proportional representation of the major habitats among locations were analysed using single factor analyses of variance on arcsin transformed data. Differences among locations and sites in the density of *Centrostephanus* were analysed using a two factor nested analysis of variance with the factor Site nested within Location. The results of these tests are reported in the text along with pairwise *a posteriori* tests (Ryan's tests with $P < 0.05$) for differences among means following significant F ratios.

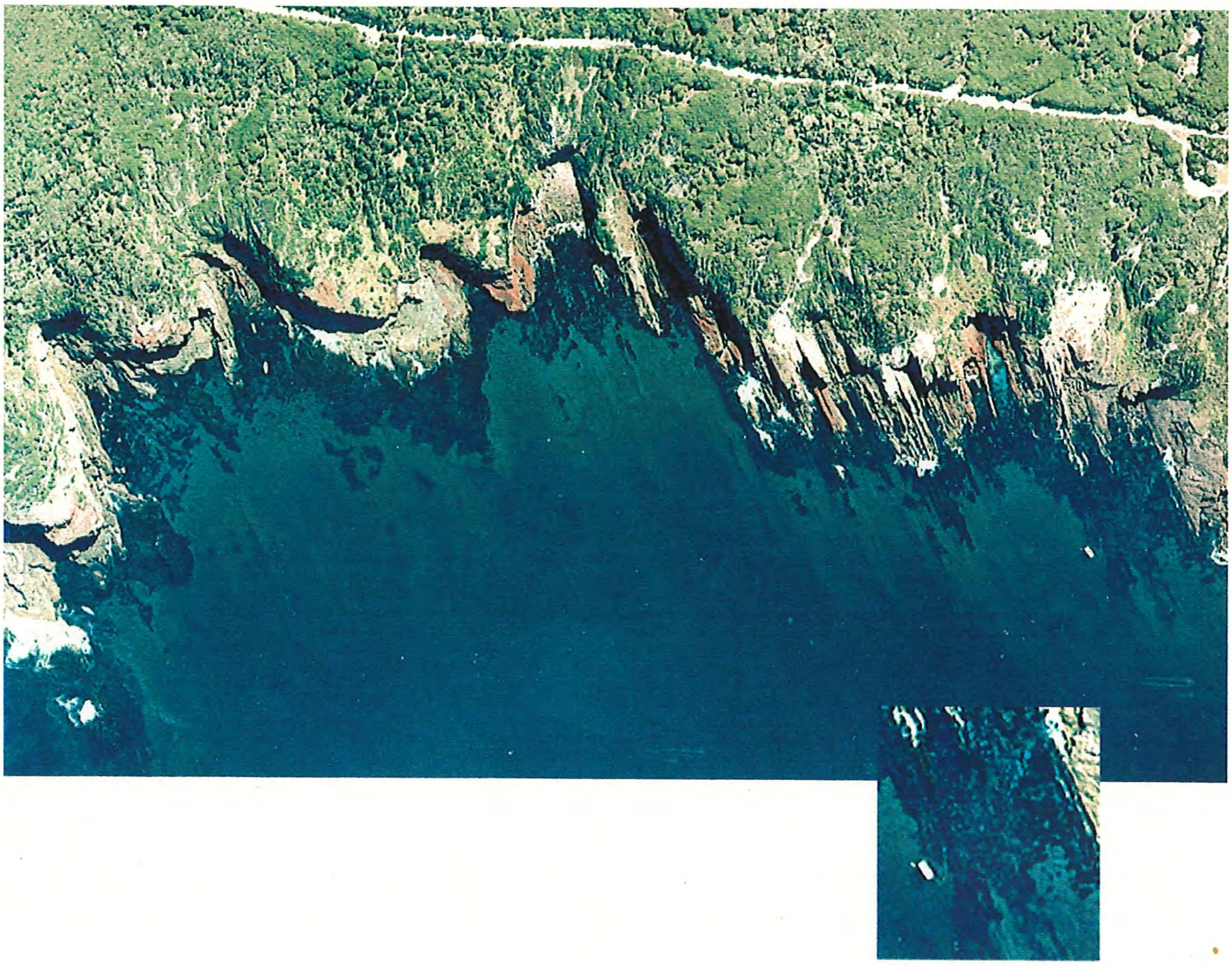


Figure 3.2. Detail of 1:8,000 photograph of Green Cape, New South Wales. The site shown is approximately Disaster Bay Site 1. Note the abalone boat with hookah hose and diver in the lower left corner (see inset for detail). The images as processed had a higher degree of resolution than is shown above.

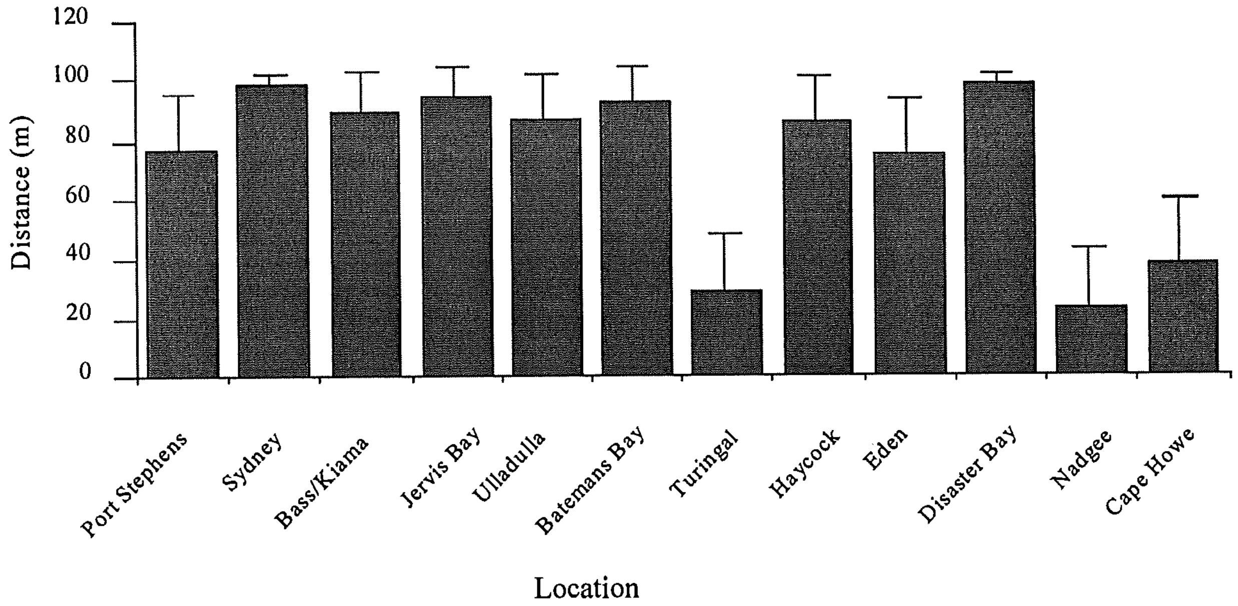


Figure 3.3. Mean proportion (+/- SE) of sites per location at which contiguous reef ended within 150 m from shore at 12 locations along the NSW coast.

3.3 Results

The 1: 8000 photographs (Figure 3.2) and subsequent scanned images provided sufficient spatial resolution to allow unambiguous maps to be drawn of subtidal habitats. Although photograph quality depended on water turbidity, depth, and sea surface conditions, there were sufficient photographs of the neccesary quality to allow the description of broad scale patterns in habitat representation. A subset of the maps derived from the aerial photographs is presented in Appendix 1.

Within the 150 m boundary used, the reef was significantly smaller at the two most southern locations (Cape Howe and Nadgee), and Turingal than any other location (Figure 3.3, $F_{(11,48)} = 12.13$, $P < 0.001$, Ryan's tests). There were no significant differences among all other locations in the area of reef within the 150 m buffer zone used (Ryan's tests). The reef gave way to sand within 150 m from shore in 60% or 36 of the 60 sites mapped. The recorded representation of habitats at these sites is, therefore, representative of all habitats found at those sites. The reef ended within 150 m of the shore in at least one site at all locations except Sydney, and Disaster Bay. At those sites where the reef extended beyond the 150 m buffer, inspection of photographs revealed that the reefs outside the 150 m buffer were comprised of a range of habitats, including extensions of the Barrens habitat. The Deep Reef, *Ecklonia* forest and Turf habitats are likely to be under-represented at these sites. At several sites at Cape Howe there were extensive offshore reefs that were isolated from those mapped inshore. The photographs indicate that there were areas of Barrens habitat on these reefs.

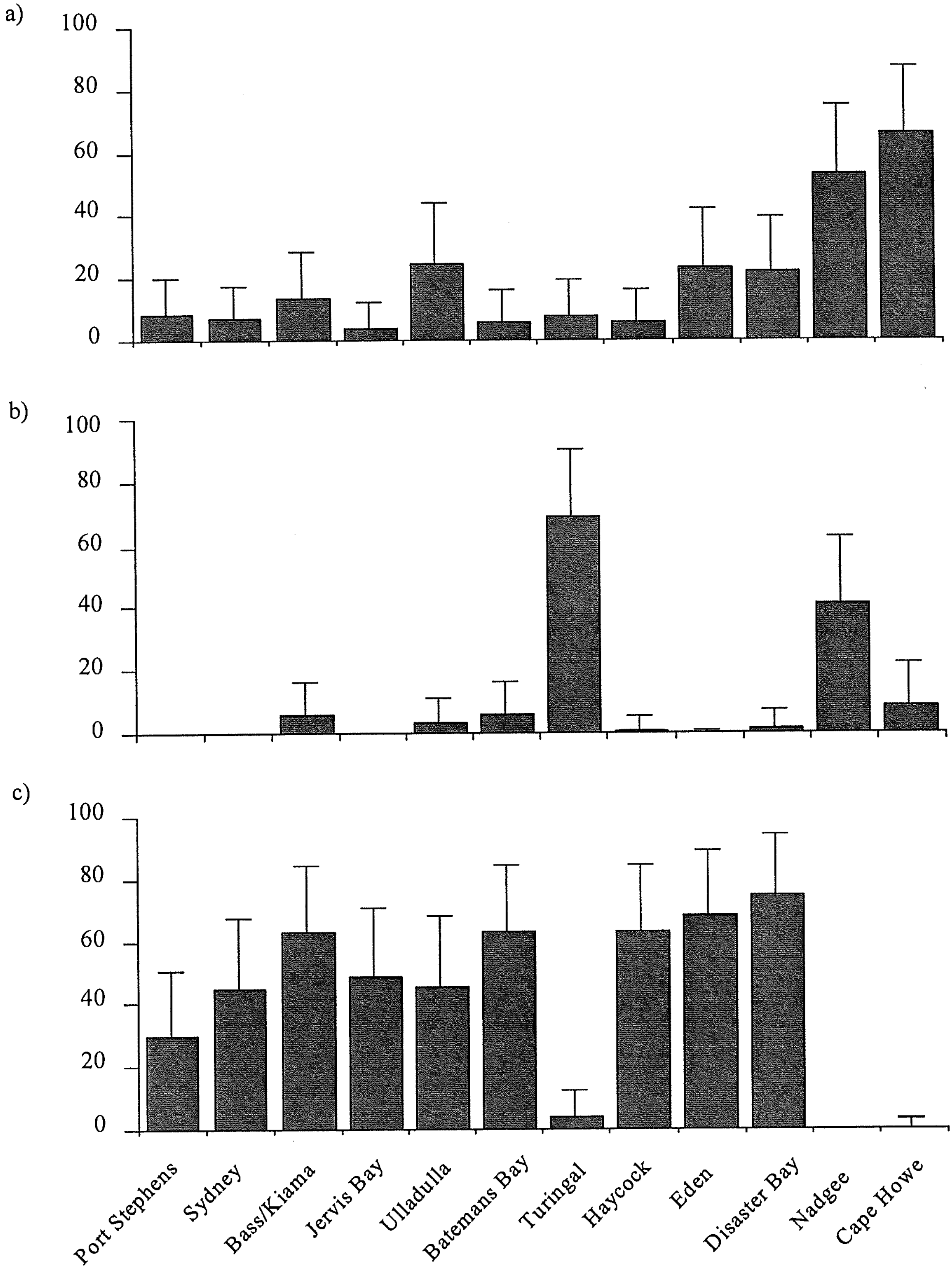


Figure 3.4 Mean percentage representation (+/- SE) of habitats on near shore reefs: (a) Fringe habitat, (b) *Phyllospora* forest, and (c) Barrens habitat at 12 locations along the NSW coast.

At Nadgee Site 5, 0.89 ha of the reef present when the photograph was taken on 9 April 1997 had been inundated with sand before 29 September 1997. Although the total area of reef at this site was small (1.35 ha), the area inundated represented 65% of the reef present approximately 6 months previously. The reef at this site, as it was at all sites at Cape Howe and Nadgee, was low relief and the presence of scoured areas and a veneer of sediment over the reef suggested these reefs were sand impacted. There was no evidence of sand inundation at any other site.

The Fringe habitat dominated the nearshore reef at Cape Howe and Nadgee (Figure 3.4a). There was a significantly greater representation of Fringe habitat at these locations than at any other location (Figure 3.4a, $F_{(11,48)} = 5.06$, $P < 0.001$, Ryan's tests). At most sites at Nadgee and Cape Howe, the Fringe habitat extended from the intertidal zone to the sand. Where this was not the case, the *Phyllospora* habitat dominated the reef, particularly at sites at Nadgee where all the reef at two sites was covered with a dense *Phyllospora comosa* forest (Figure 3.4b). *Phyllospora* was also common within the Fringe habitat, along with *Pyura* spp., turfing algae and *Ecklonia radiata*. *Ecklonia* plants found at Nadgee and Cape Howe had an unusual growth form (for NSW) in that the primary laminae and laterals were elongate and smooth (see Figure 3.120A, Womersley 1987). When plotted by site and latitude there was a clear decline in the range of observed representations of Fringe habitat with decreasing latitude (Figure 3.5a). The entire range of estimated representation was observed at latitudes greater than 36° South. In contrast, the Fringe habitat covered less than 20% of the sites mapped between 32 and 33° South.

The *Phyllospora* habitat was most represented at Turingal and Nadgee where it covered a mean of 62% (SE = 17%) and 40% (SE = 21%) of the reefs respectively (Figure 3.4b). This representation was significantly greater than at any other locations ($F_{(11,48)} = 7.13$, $P < 0.001$, Ryan's tests). There was no significant difference among all other locations in the percentage representation of the *Phyllospora* habitat (Ryan's tests). This habitat was absent from the two most northern locations although individual plants were observed at one site in Port Stephens and two sites in Sydney. These latter locations are close to the northern limit of distribution for this species. When all 60 sites were plotted as a function of latitude (Figure 3.5b), there was a clear decline in the representation of the *Phyllospora* habitat north of 36° South.

The Barrens habitat covered an estimated 50% (SE = 3.9%) of nearshore reefs between Port Stephens and Wonboyn Beach, New South Wales. Coverage was greatest in Disaster Bay where 68% (SE = 6.7%) of nearshore reef was Barrens Habitat (e.g. Figure 3.2). Although there were extensive areas of Deep Reef and Turf Habitat in deeper water, further from shore at sites in Disaster Bay (N.L. Andrew pers. obs), these were beyond the mapped reef. At the most northern location, Port Stephens, Barrens covered 30% (SE = 15%) of the nearshore reef. South of Wonboyn Beach, at Nadgee and Cape Howe, only 1% (SE = 0.3%) of nearshore reefs were Barrens habitat (Figure 3.4c). These locations, and Turingal, had significantly less Barrens habitat than any other location ($F_{(11,48)} = 7.79$, $P < 0.001$, Ryan's tests).

With the exception of Turingal, Nadgee and Cape Howe, where the nearshore reefs were small, there was a trend toward increasing mean representation of Barrens habitat with increasing latitude (Figure 3.4c).

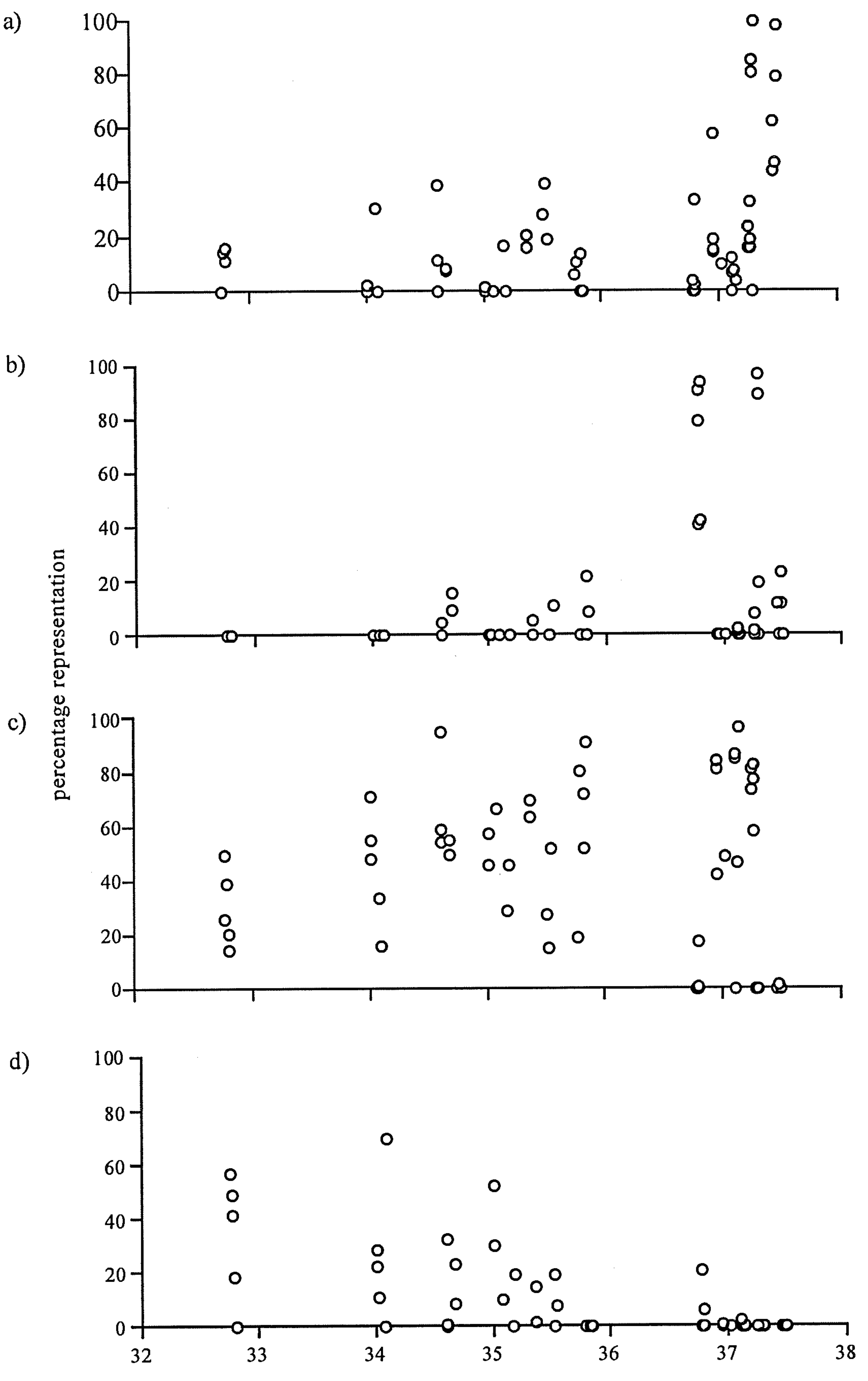


Figure 3.5 Scattergrams of percentage representation of: (a) Fringe habitat, (b) *Phyllospora* forest, (c) Barrens habitat and (d) *Eklonia* forest as a function of latitude at 60 sites along the NSW coast.

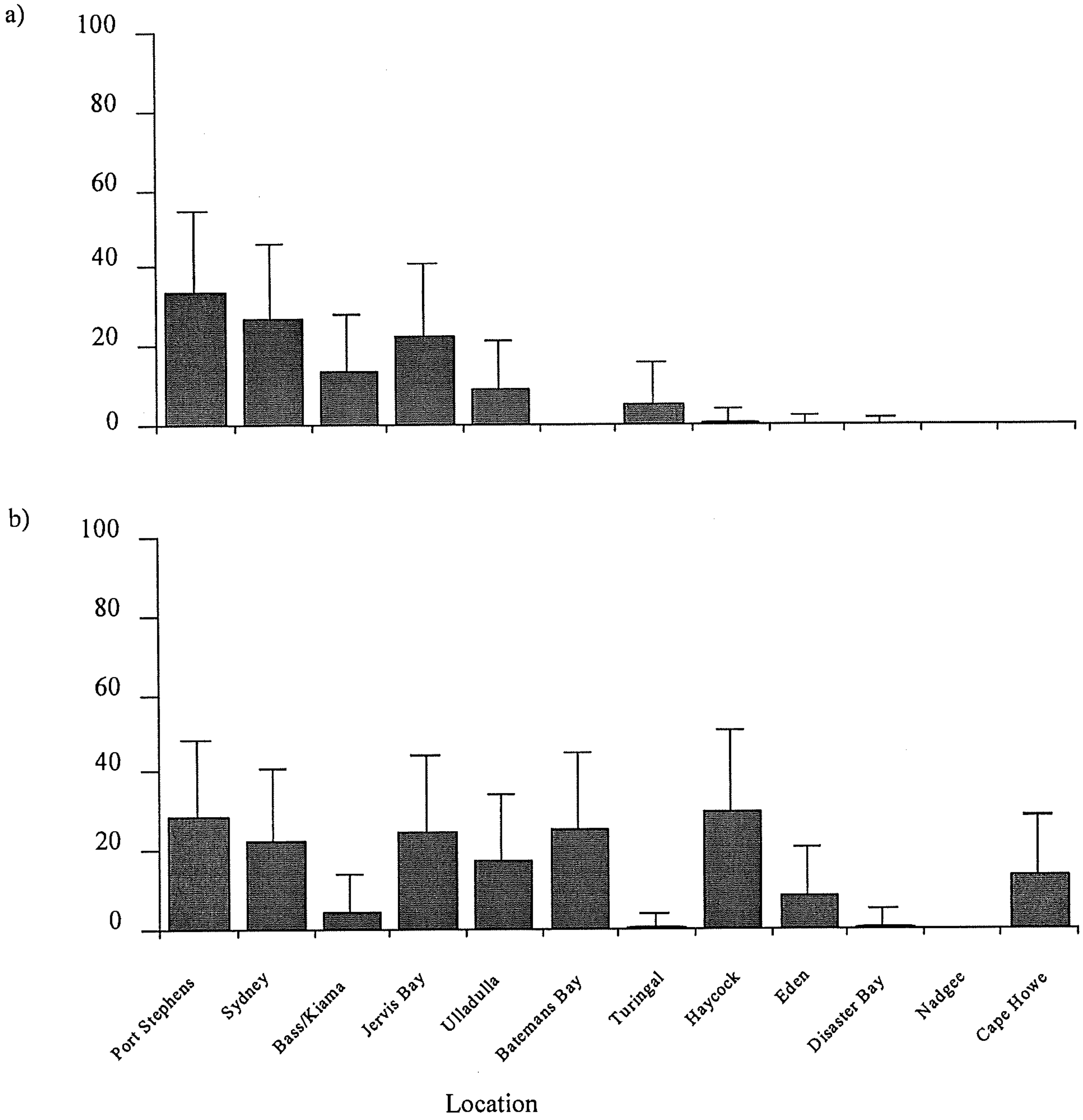


Figure 3.6 Mean representation (+/- SE) of habitats on nearshore reefs: (a) Ecklonia forest, (b) Turf habitat at 12 locations along the NSW coast.

However, when all 60 sites were plotted against latitude, there was no significant correlation between latitude and representation of the Barrens habitat (Figure 3.5c). There was a significant positive correlation between the proportional representation of Barrens habitat and the area of nearshore reef ($P<0.05$).

The maximum observed representation of *Ecklonia* forest declined at sites with increasing latitude (Figure 3.6a). At sites between 32 and 34° South, a wide range of coverage of *Ecklonia* forest was recorded, however, in the far south of the state, large areas of canopy-forming forest were not recorded during this study (Figure 3.5d). There was significantly greater representation of the *Ecklonia* forest habitat at Port Stephens than any other location ($F_{(11,48)} = 5.06$, $P < 0.001$, Ryan's tests). *Ecklonia* forests were absent from all mapped sites at Nadgee and Cape Howe and from 4 of the 5 sites at each of Batemans Bay, Disaster Bay and Haycock Peninsula. Although *Ecklonia* plants were present at all locations and sites, densities were usually not sufficient to form a canopy (N.L. Andrew per obs). Such areas of reef with sparse *Ecklonia* were categorised as being Turf habitat. Extensive areas of *Ecklonia* forest may be found inside South Head in Twofold Bay, however, these more sheltered sites were not mapped in the present study.

There were no significant differences among locations in the representation of the Turf habitat (Figure 3.6b, $F_{(11,48)} = 1.27$, ns). Although the Turf habitat was extensive at some sites, there was considerable variability among sites within locations. Large brown algae such as *Cystophora monoliformis*, *Ecklonia radiata* and *Sargassum* spp. were common in patches of Turf habitat. By definition, *Phyllospora comosa* and *Pyura* spp. were absent.

In addition to those described above, two additional habitats were recorded but generally accounted for a minor proportion of the coverage of nearshore reefs. The *Pyura* habitat was recorded at 3 of the 60 sites (at Bass Point, Jervis Bay, and Turingal). This habitat was most represented at Turingal where it accounted for 57% of the small area of reef present at one site. At Bass Jetty (Bass Point) and Mary Cove (Jervis Bay) it accounted for less than 5% of the reef area. The *Pyura* habitat dominates many reefs between Nadgee and Wonboyn Beach, at sites not mapped during the present study (N.L. Andrew pers. obs.).

The second additional habitat, *Durvillaea*, was found only in the far south of the state. The bull kelp *Durvillaea potatorum* was present at most sites south of Turingal and was found either at high densities in a narrow band in the immediate subtidal and low intertidal zone or as scattered plants on the tops of ridges and places with the greatest wave exposure. *Durvillaea* was sufficiently dense to form a canopy at sites in Turingal, Disaster Bay, Nadgee and Cape Howe. At all sites this habitat accounted for less than 2% of the total nearshore reef area.

The abundance and length-frequency structure of *Centrostephanus* was estimated at nine of the mapped locations (a total of 27 sites). At these locations, *Centrostephanus* were most abundant at Bass Point/ Kiama where, at several sites, densities exceeded 75 per 10 m^2 (Figure 3.7). There were significant differences in the density of *Centrostephanus* among locations ($F_{(8, 18)} = 3.91$, $p < 0.01$) and sites within locations ($F_{(18,378)} = 3.91$, $p < 0.001$). There was no clear north-south trend in mean density among locations or sites (Figure 3.7), nor was there a significant correlation between latitude the mean density of *Centrostephanus* at sites ($r_{(25)} = 0.11$, ns).

Urchins were observed outside shelters, such under boulders or gutters, at sites at Bass Point/Kiama, Ulladulla, Haycock, Eden and Disaster Bay (Figure 3.7). These urchins were commonly but not always aggregated in tight groups. These patterns differ markedly from those found on reefs at Sydney where the Hawkesbury sandstone produces large undercut platforms. Very few urchins at the Sydney sites were observed to be fully exposed on open reef. Exposed urchins were only observed at sites with urchin densities greater 30 per 10 m^2 but not all sites with such densities had exposed urchins (Figure 3.7). The relationship between density of urchins, the area of Barrens habitat they are in, and the type of rock at the site remains poorly understood.

The size-structures of *Centrostephanus* at all sites were dominated by one or, less often, two modes. Most usually, the size distributions were dominated by a single mode between 65 and 85 mm T.D. Of the 27 sites sampled, divergent examples only are presented to indicate the range of length-structures described (Figure 3.8, see Appendix 2 for the full series). At Anna Bay Site 1, Port Stephens, the population was dominated by large individuals, with a modal size class of 90 mm TD (Figure 3.8a) whereas the population at the adjacent site 2 was comprised of much smaller urchins, with a modal size of 65 mm TD (Figure 3.8b). In contrast, at other locations sampled, such as Sydney and Eden, there was little difference among sites in population size-structure (Appendix 2). At Sydney Site 1, there was evidence of two modes in the size-structure, at 50 and 80 mm TD (Figure 3.8c). In contrast, Kiama Site 1 and Batemans Bay Site 1 were comprised only of urchins between 40 and 70 mm TD (Figure 3.8d, e). The proportion of urchins near the modal size varied among sites. For example, at Sydney, the distributions were relatively flat, with a wide range of sizes of urchins present at all sites (Appendix 2b or Figure 3.8c) whereas other sites, such as Eden Site 3 had relatively small ranges of sizes of urchins (Figure 3.8).

For those sites with uni-modal size-frequency distributions (N = 24, Appendix 2), there was a significant negative correlation the mean density of sea urchins at a site and the modal size of individuals at that site ($r_{(22)} = 0.53$, $P < 0.05$). The smallest *Centrostephanus* observed were between 40 and 45 mm T.D.. Urchins less than 50 mm TD were found at all three sites in Sydney but, rarely, at other sites. At some sites, such as Haycock site 2 and Port Stephens site 2 urchins less than 65 mm TD were rare (Appendix 2).

3.4 Discussion

The results presented provide the first estimates of habitat representation across the geographical range of the NSW abalone fishery. The predominance of the Barrens habitat on reefs throughout the range of the NSW abalone fishery and the known negative association between these species (Shepherd 1973, Andrew and Underwood 1992, Chapter 4) suggests that sea urchins limit the productivity of the abalone fishery. The development of a sea urchin roe fishery offers the potential to enhance the abalone fishery by reducing the abundance of sea urchins on reefs. Although such a fishery is likely to concentrate on sea urchins from the Fringe habitat in the first instance (Chapter 5), the expansion of the fishery into other habitats may follow. This expansion is likely to come about through more sophisticated management such as rotational closures and possibly the enhancement of roe quality through control of urchin densities in the Barrens habitat (see Chapter 5 for further details of possible management regimes). Small scale experiments (Andrew and Underwood 1992, Andrew 1993) suggest, however, that there is likely to be a non-linear relationship between the abundance of *Centrostephanus* and the area of Barrens on a reef. As densities decline we hypothesise that the patches of Barrens will fragment and that

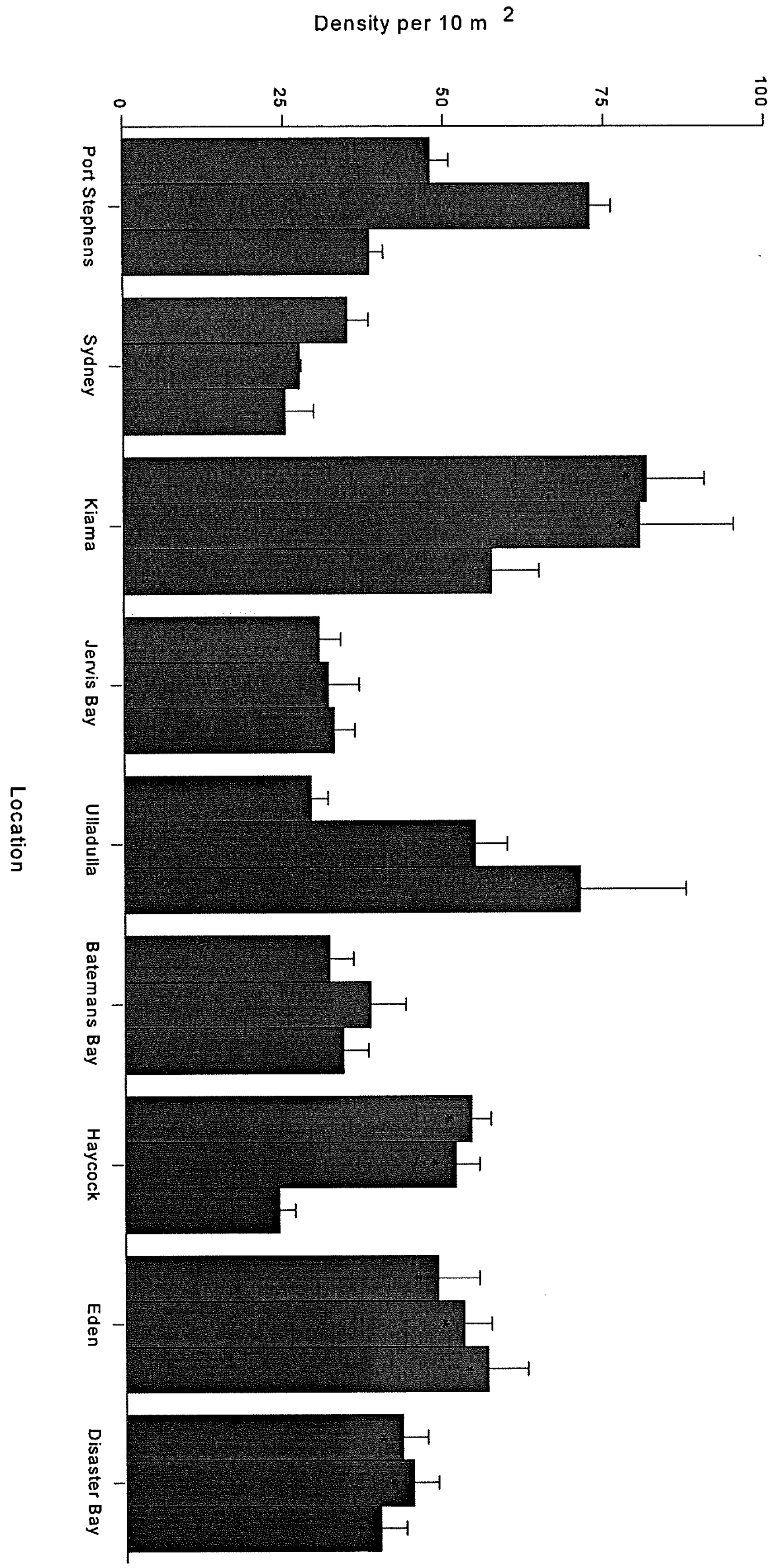


Figure 3.7 Mean density (+/- SE) of Centrostephanus per 10 m ^{2} at 27 sites at 9 locations along the NSW coast. Asterisks indicate those sites at which urchins were observed outside shelters during the day.

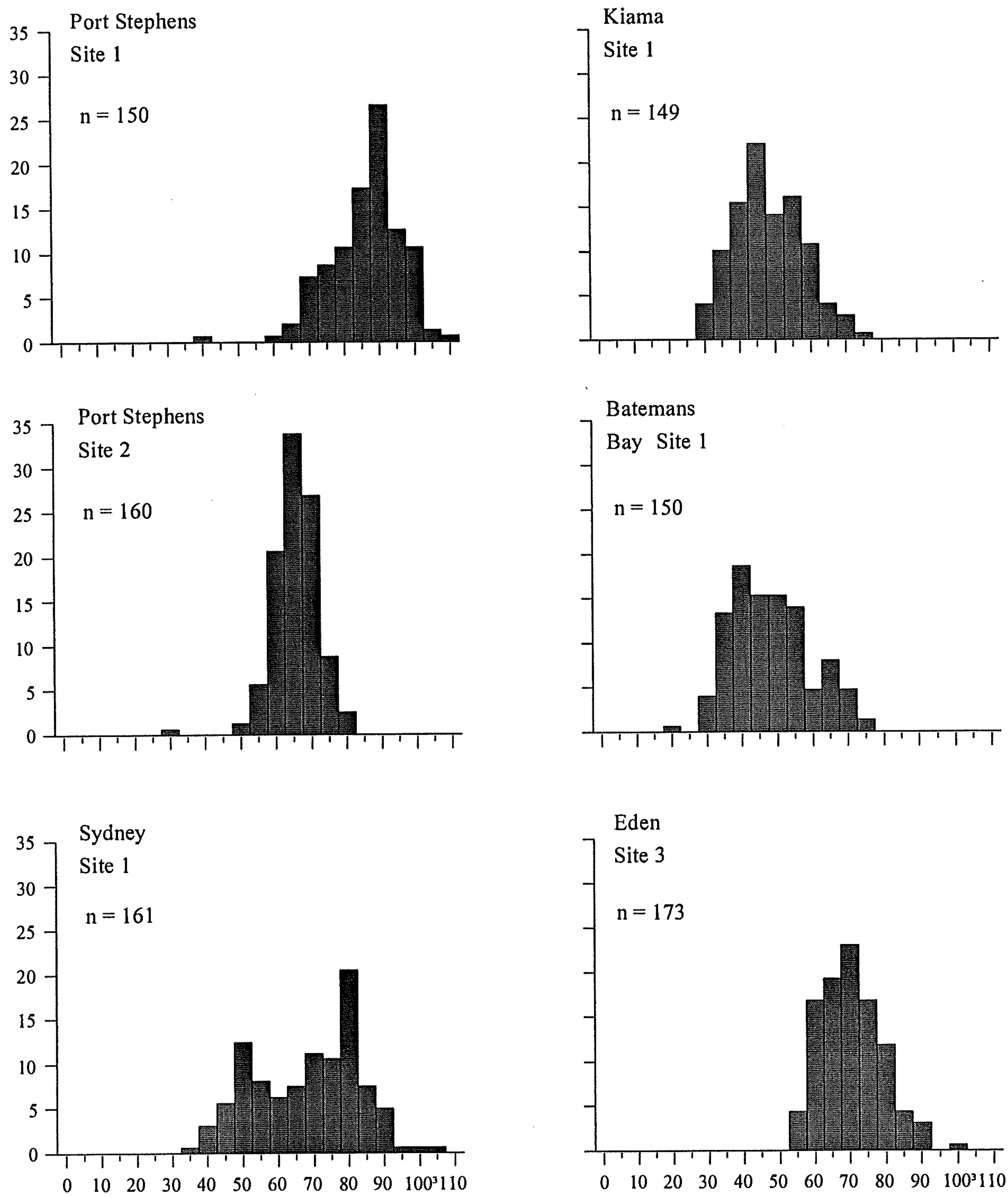


Figure 3.8 Size frequency distributions of Centrostephanus at six sites in New South Wales. Sample sizes as indicated on each graph.

densities within these patches will decline at a slower rate than the overall abundance of urchins at a site. The implications of this on the quality of roe in the remaining urchins is unknown.

The results of this study broadly support the habitat descriptions and estimates of habitat representation reported by Underwood et al. (1991). The addition of the *Durvillaea* habitat reflected the extension of the study to more southern reefs. The *Pyura* habitat was recorded only at the two most northern sites, at Southwest Rocks and Charlotte Head in Underwood et al (1991). In the present study this habitat was found at sites in the far south of the state. In both Underwood et al. and the present study, the *Pyura* habitat was strongly associated with sites in which the reef was small. The Turf habitat was found to be the second most common following the Barrens habitat. The composition of the habitat varied considerably, depending on location but was always characterised by a high percentage cover of articulated coralline algae.

Few previous estimates of habitat representation on shallow subtidal reefs are available. Existing aerial photographs proved to be inadequate to allow comparisons to be made. The only other estimates available are the transects done in 1988 and reported in Underwood et al. (1991). In that study, the sites mapped at Merimbula are comparable to those mapped at Haycock Peninsula in the present study. Comparison between these two estimates is made difficult by the fact that sampling in Underwood were limited to the first 100 m from shore, which may the comparison with 150 m mapped in the present study. This bias would underestimate the representation of the Barrens habitat in 1988 more than the present study. Nevertheless, inspection of Figure 2 in Underwood et al. (1991) suggests that between 50 and 70% of reefs were Barrens in 1988, compared to 68% (SE = 8.7%) representation at Haycock peninsula in 1996-7. Similarly, at Cape Banks, in Sydney the estimated proportion of Barrens in 1988 was 30-40%, compared with the single estimate in the present study of 53%. These comparisons, although they should be taken with some caution, suggests that there has not been major changes in the representation of Barrens at these sites.

The differing methods used to quantify habitat representation in Underwood et al. (1991) and the present study offer insights into the patchy nature of habitats on NSW nearshore reefs and the limitations of both methods, which differ in grain, extent and intensity of sampling (Wiens 1989, 1990 and references therein). Sampling grain refers to the lower limit of resolution in the sampling. In the aerial photographs, the sampling grain size was an individual pixel in the digitised image, which represented approximately 0.5 m^2 of reef. In practice, the smallest 'patch' distinguished in creating the maps was larger, because of uncertainties in interpreting the images and in locating patches on the map during ground truthing. These operational facts meant that very small-scale heterogeneity (<2 m^2) was averaged out. The impact of this on estimates of area obtained varied among locations and habitats but in general was minor because of the size of patches and the sharpness of boundaries between the Barrens habitat and those dominated by foliose algae (e.g. Figure 3.2 and Andrew 1993, 1994). The mosaic of Fringe, *Phyllospora* and Turf habitats in the immediate subtidal made detecting patches more difficult and heterogeneity less than approximately 2 m^2 would have been averaged. It should be noted that the information content of the original photographs was greater than was used in the present study.

Digitising the original negatives at higher resolution and processing the images on larger and higher resolution computer screens would allow very small scale heterogeneity to be described.

The transect-based sampling was much more finely grained in that it could quantify habitat down to 10 cm along the tape measure. However, this sampling was both less extensive (covered a smaller area - 80 m of coast) and less intense (the actual area quantified was small - the points under four tape measures < 100 m long). Given the complex way different habitats were arranged in space (Figure 3.2, Appendix 2), running four transects normal to the shore may have meant that the sampling was not sufficiently intense to reliably quantify habitat representation. For example, the reefs were surveyed twice, in summer and winter 1988. Large differences were observed in the proportional representation of some habitats both among transects within surveys and between surveys separated by 4-5 months. It is improbable that such large changes would occur over this time period because the large brown algae and sea urchins that define these habitats recruit and grow to a size that would cause habitat change over much longer time periods. The observed differences are more likely to reflect artefacts associated with laying transects out perpendicular to the shore in such reticulated reefs.

The quantification of habitats in this study was limited to 150 m from the nearest point in the intertidal zone. This constraint was caused by limitations in the capacity of the aerial photography to penetrate water. Water clarity, sea surface conditions and depth limited the application of the technique to relatively shallow depths. In considering the representation of habitats in the subtidal zone in New South Wales, the Deep Reef habitat will be under-represented at the 40% sites with rocky reef beyond the frame of the present study. The large differences in representation of the Barrens habitat at Merimbula reported in the present study and in Underwood et al. (1991) suggest either a dramatic change in one habitat type only or highlights an inconsistency in the definition of this habitat. Because sea urchins and crustose coralline algae may be abundant within the Deep Reef habitat (Underwood et al 1991), the difference may simply be an artefact of interpretation of the differences between Deep Reef and Barrens habitats.

The diversity of size structures and abundances of sea urchins observed in the patches of Barrens mapped provides a framework for testing hypotheses about the link between the demography of *Centrostephanus* and habitat structure. There were clear differences in the abundance and size-structure of *Centrostephanus* among sites and locations. The implications of these differences to the long-term persistence of the Barrens habitat are unclear. If the size-structures are indicative of age-structures, there appears to be large differences in growth and/or recruitment among locations and sites. In particular, the apparent lack of small sea urchins (< 40 mm TD and 2 years old, Andrew 1991) at all but a few sites suggests either sporadic recruitment into populations of long-lived urchins or the potential for a greater degree of dynamism in habitat structure than previously thought (e.g. Andrew and Underwood 1989, Andrew 1993).

In contrast to studies around Sydney (e.g. Fletcher 1987, Andrew and Underwood 1989, Andrew 1993) *Centrostephanus* were observed in the open during daylight at some locations. Although this was most prevalent in the south of the state, large aggregations were also observed in the open at Bass Point.

Although data were not collected on the proportions of sea urchins in the open at each site, it appears that they were greatest in areas with high densities and or relatively little shelter (N.L Andrew pers. obs.). These observations suggest that the influence of rock-type noted by Andrew and Underwood (1989) may be extended to patterns of dispersion and shelter-dependence. There was no evidence, at least from their daytime patterns of distribution and dispersion, that these aggregations were mobile and that the urchins were foraging outside the Barrens habitat. The long-term consequences of high densities and the occurrence of aggregations of urchins in the open remain unknown. The maps produced as part of the present studies will however, provide a basis for assessing their consequences to the persistence of patches of Barren habitat, and their size and location on the reef. Further, the maps will allow different questions to asked concerning the importance of urchin behaviour to the nature and persistence of adjacent patches of habitat and the scale-dependence or otherwise of sea urchin herbivory (Dayton and Tegner 1984, Levin 1992, Aronson 1994 and references therein).

4. Influence of Habitat on the Abundance of Abalone

Objective 3: To describe the influence of habitat type on the abundance of abalone.

Our results suggest the removal of sea urchins and the consequent change in habitat type from the Barrens habitat type to one dominated by a range of foliose algae causes an order of magnitude increase in the density of abalone. It is, therefore, possible to enhance the densities of abalone in NSW by removing *Centrostephanus*. The mechanism underlying this effect is unclear because removing sea urchins causes large changes in habitat structure as well as increasing available shelter. There is some evidence that densities may further be enhanced by the addition of adult abalone. However, this latter effect is swamped by the impact of removing sea urchins. The impact of removing sea urchins appeared to be consistent across a range of spatial scales and does not appear to be dependent on a particular period of the year.

4.1 Introduction

In New South Wales, the sea urchin *Centrostephanus rodgersii* is the dominant herbivore on shallow rocky reefs. Its herbivory is responsible for the creation and maintenance of a major habitat, commonly termed Barrens or 'white rock areas' (Fletcher 1987, Andrew 1993, Andrew and Underwood 1989, 1992, 1993, Underwood et al. 1991). Abalone are rare in this habitat (Shepherd 1973, Andrew and Underwood 1992, 1993, Underwood et al. 1991, McShane 1991, Andrew 1993). Commercial divers have claimed that sea urchins exclude abalone from the Barrens habitat and, more recently, that the area of reef covered by this habitat is increasing, at the expense of productive areas of reef. This observation and explanation for the lack of abalone in the Barrens habitat was first made 25 years ago by Shepherd (1973) and, prior to the present study has not been tested by experiment on an appropriately large scale, or in areas in the south of the state where abalone densities are greatest.

Although there is evidence that sea urchins have a negative impact on the abundance of abalone, the underlying mechanism is not clear. Several hypotheses have been proposed, including competition for shelter (Shepherd 1973, see also Lowry and Pearse 1973, for other species of abalone and sea urchin) and incidental predation of juvenile abalone by large sea urchins (Naylor and McShane 1997a). The relationship is further complicated by the fact that abalone are thought to recruit in greatest numbers on crustose coralline algae (Morse and Morse 1984, Shepherd and Turner 1985, Naylor and McShane 1997b) which are most abundant in the Barrens habitat (Andrew and Underwood 1989, Underwood et al. 1991). These seemingly contradictory hypotheses and results suggest that the effects of sea urchins on the abundance of abalone are likely to depend on the size of abalone, the scale at which observations are made, and a range of auxiliary factors such as sea urchin density and the abundance of filamentous and foliose algae.

Possibly interacting with the broader impacts of sea urchins are the processes that limit the availability and subsequent recruitment of juvenile abalone. There is evidence that the processes controlling the numbers of recruiting abalone operate at relatively small spatial scales. Studies in Tasmania and Victoria have provided preliminary evidence for localised recruitment in blacklip (Prince et al. 1988a, b, McShane 1995, Shepherd and Partington 1995). In contrast, no evidence for localised stock-recruitment relationships have been found in other species (e.g. Shepherd et al. 1990, Tegner et al. 1992). If populations of blacklip abalone replenish themselves with relatively little input from distant reefs, and further, if sea urchins are displacing abalone on a large scale, then abalone may have a limited ability to re-colonise those areas. If this can be demonstrated to be the case then there are obvious implications for management. Perhaps most importantly, the appropriate scale of management may be smaller than normally considered for marine fisheries.

In this Chapter we describe the results of an experiment which tests several hypotheses concerning the effects of sea urchins (through changes in habitat structure) on the abundance of abalone and secondly the results of larger-scale removals of sea urchins done in conjunction with commercial abalone diver. Finally, we outlines changes to the regulations governing the harvest of sea urchins that were made as a consequence of these experiments.

4.2 Small-scale sea urchin removals

4.2.1 Methods

In this first experiment, the removal of sea urchins was done in conjunction with several related treatments to test hypotheses about the timing of removal (or harvest) and the scale at which abalone populations replenish themselves by recruitment. Specifically, we asked the following questions:

(1) Does the removal of sea urchins increase the abundance of abalone?

(2) Does the timing of removal influence any effect of removing sea urchins?

(3) Do removals need to be maintained or is it sufficient to remove sea urchins only once?

(4) Does the introduction of conspecific adults increase the densities of recruit abalone?

The hypotheses outlined above were tested in a multi-factorial experiment. The design contained three main factors (Table 4.1):

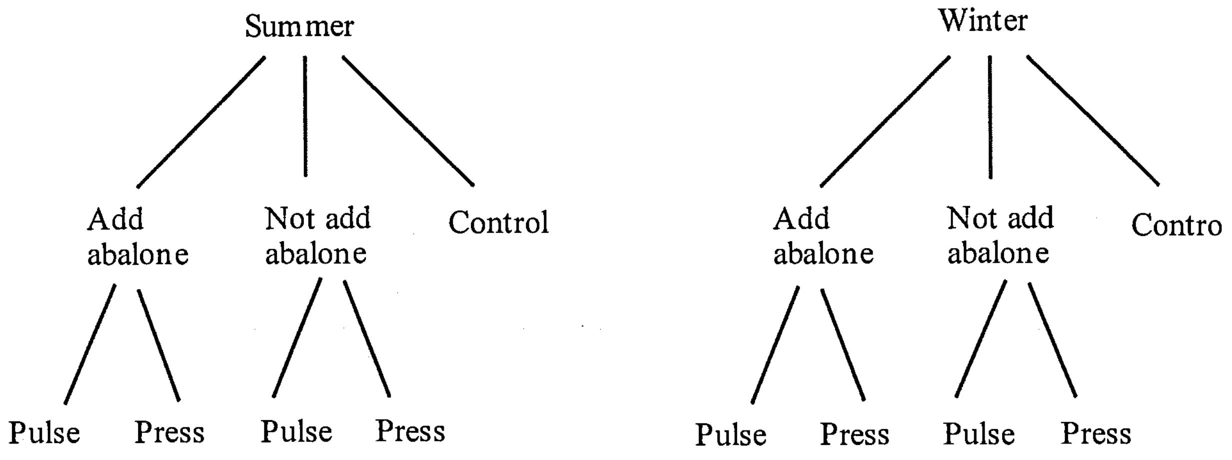
(1) **Sea urchins.** This factor had three treatments (press, pulse and control). Urchins were removed from areas of reef and the areas either maintained free of all sea urchins (press) or *Centrostephanus* were only removed initially (pulse). In pulse treatments, the areas were totally cleared initially, and then only

cleared around the perimeter in later sampling periods to prevent urchins moving in from other areas. In this way, recolonisation was only possible through recruitment of juveniles. In both pulse and press treatments, the clearances were achieved in several episodes over several months as previously cryptic sea urchins became obvious after the initial clearance. Control areas, in which *Centrostephanus* densities were not manipulated, were sampled as for the manipulated sites. These controls were not orthogonal with the core block of factors Abalone, Sea Urchin and Season (see below).

(2) **Abalone**. This factor had two levels (added and not added). Reproductively mature abalone were added to half the sites to determine whether the addition of broodstock increased the densities of recruits. There were two processes that might be operating to produce differences between these treatments. First, there may be a relationship between the stock of abalone in a site and the supply of recruits to that site. This may possible because of the hypothesised restricted dispersal of larval abalone. Second, the supply of abalone to the treatments could be similar, but larvae either choose to recruit, or survive better, in the sites with more adult abalone. Unfortunately, these different mechanisms can not be distinguished in this design. Nevertheless, the consequences of such interactions are the same for the management of the fishery. That is, differences in the abundance of recruits among the treatments suggests a relationship between stock and recruitment at a small spatial scale.

(3) **Season**. The removals of sea urchins and addition of mature abalone was repeated in winter and summer. The time of year was hypothesised to have an influence on the species composition of algae that would colonise the sites. For example, algae such as *Ecklonia radiata* have marked seasonal peaks in recruitment and we hypothesised that the species of large brown algae that recruited to the clearances may influence colonisation and recruitment of juvenile abalone. Technically, the factor Season is unreplicated because we had only one 'example' of each season in the design. We acknowledge this fact but the replication of seasons was beyond the capacity of this study. The interpretation of the (lack of) differences among seasons should be interpreted with this limitation in mind.

Table 4.1 Design of the experiment showing the levels of the three main effects, their interactions and the non-orthogonal controls. A fourth factor, Gutter was nested within the three factor interaction term.



The experiment was asymmetric, with a core orthogonal component of Sea Urchin (press and pulse), Abalone (added and not added) and Season (summer and winter), plus the non-orthogonal controls. In addition , a fourth factor, Gutter, was nested within the three factor interaction term. In October-November 1993, 30 replicate gutters were selected for the experiment on the north and south sides of Twofold Bay, southern New South Wales. Of these, 15 replicates were randomly selected and sampled prior to sea urchins being removed from 12. The remaining 15 replicates were randomly assigned to treatments in May 1994. These treatments were manipulated in the same way as the summer treatments. Six months after initiation of the sea urchin treatments, 100 reproductively mature abalone were added to each of 6 gutters. Six months after these additions, a further 50 reproductively mature abalone were added to each of the gutters from which sea urchins had been removed.

Three replicate gutters were used per treatment combination. Each gutter was an area of reef of at least 1000 m^2 separated by at least 20 m and/or a natural partition in the reef, most commonly a rock wall that reached the surface of the water. The gutters were bordered inshore by Fringe habitat (Underwood et al. 1991, Chapter 3) and the intertidal zone. The original boundary between the Barrens and Fringe habitats remained clear for the duration of the experiment so estimates of abundance and percentage cover were not confounded by sampling outside the original area. On the deeper margins of each replicate there were contiguous areas of Barrens habitat and the clearances were made to natural breaks in the habitat, such as areas of sand.

After the initial set up, sampling and maintenance of removals was repeated in August 1994 (summer treatments only), November 1994 (summer and winter), February 1995 (summer only), May 1995 (summer and winter), August 1995 (winter only) and November 1995 (summer and winter), May 1996 (summer and winter) and July 1997 (Summer and winter).

The experiment was analysed for differences among treatments at the end of the experiment, in July 1997. Transformations were used as required to reduce heterogeneity of variance and are indicated in the appropriate tables. An a priori decision was made to pool non-significant interaction terms in the analyses following the guidelines in Winer et al. (1992). The data are presented graphically as subsets of the full treatment combinations based on the analyses. In addition to these analyses, differences in mean density of small abalone were compared in May 1995 after the first recruitment season following the manipulations.

In each replicate, sea urchins (*Centrostephanus rodgersii*), abalone (*Haliotis rubra*), turban shells (*Turbo torquatus*) and top shells (*Australium tentiforme*) were counted in ten 10 x 1 m randomly placed transects. Only data for sea urchins and abalone will be presented in this report. Abalone were divided into 3 categories: small (< 65 mm), medium (65-114 mm) and large (>115 mm). The transects were sampled by divers swimming slowly along the tape measure searching for the species sampled, but not overturning boulders or otherwise disturbing the reef structure.

Percentage cover of algae and sessile invertebrates were estimated by intersection with 10 random points (random point counts or RPCs, Foster 1975) in each of five throws of a RPC bar in each transect as above. Estimates of percentage cover from these 50 point estimates were pooled for each transect. The type of substratum under each knot was recorded as primary cover. Secondary and tertiary cover were recorded if there were foliose algae above the point of contact with the knot.

Between May and August 1995 there was a mass mortality of all echinoderms and molluscs in six adjacent replicates on the south side of Twofold Bay. All but a few abalone, limpets, and sea urchins had died along a strip of coastline of approximately 200 m. The boundary of the affected area appeared abrupt as replicate gutters on either side of the area were not effected. The cause of the mortalities remains unknown. The impact of this event on the experiment was to dramatically reduce densities of abalone and sea urchins in six of the 30 replicate gutters. Algae did not appear to be effected. The effected replicates were spread across six of the 10 treatment combinations (no combination had more than one effected replicate). The impact of these changes depended on treatment - those with highest densities of abalone were differentially impacted, as were the controls for sea urchins. These mortalities therefore would have biased the results of the experiment based on the apparently mysterious location of the mass mortality. For this reason, these replicates were removed from the data set after May 1995. For the analyses of difference in July 1997, the lost replicate in each treatment combination was replaced with the mean of the cell and six degrees of freedom subtracted from the appropriate denominator degrees of freedom.

4.2.2 Results

Sea urchins

After the initial clearances in November 1993 and May 1994 there was relatively little immigration by urchins into the cleared areas for 18 months after removals. In July 1997 there had been recruitment of *Centrostephanus* into the pulse treatments, but the densities of sea urchins remained at low levels in both the press and pulse treatments (Figure 4.1a). At the end of the experiment, in July 1997, there were significantly more *Centrostephanus* in the pulse treatments than in those in which the removals had been maintained (Table 4.2). The mean density of urchins in the pulse treatment in July 1997 was 15.1 per 10 m^2 (SE = 1.67, n = 18, pooled across all other factors). The density of sea urchins remained relatively constant through time in the control sites (Figure 4.1a) except for a sharp decline in November. The season during which the clearances were done had little apparent impact on densities of sea urchins in the experiment in July 1997 (Table 4.2). There were significant differences among replicate gutters in the density of sea urchins in July 1997 (Table 4.2).

Algae

In the gutters from which sea urchins had been removed, there was a large increase in the percentage cover of non-crustose algae (Figure 4.1b, c). Principally these were articulated corallines, such as *Amphiroa* spp. and *Corallina* spp., and foliose algae. Following the removal of sea urchins, there was an immediate increase in the percentage cover of filamentous species. Subsequently, there were large differences among gutters in the relative dominance of turfing and foliose algae. These differences were reflected in the significant effects of Gutters and three factor interaction terms in the analyses of differences for these groups of species (see below).

Following the removal of sea urchins, there was a steady increase in the combined percentage cover of foliose algae in all treatments (Figure 4.1b, c). There was a significant difference in cover between the controls and all urchin removal treatments in July 1997 (Table 4.2). In July 1997, there was no evidence that the increase in density of urchins in the pulse treatments reduced the density of foliose algae (Figure 4.1b, c, Table 4.2). None of the other main effects or their interactions were significant when analysed (Table 4.2). Data are plotted separately by Summer and Winter for comparison with other species of

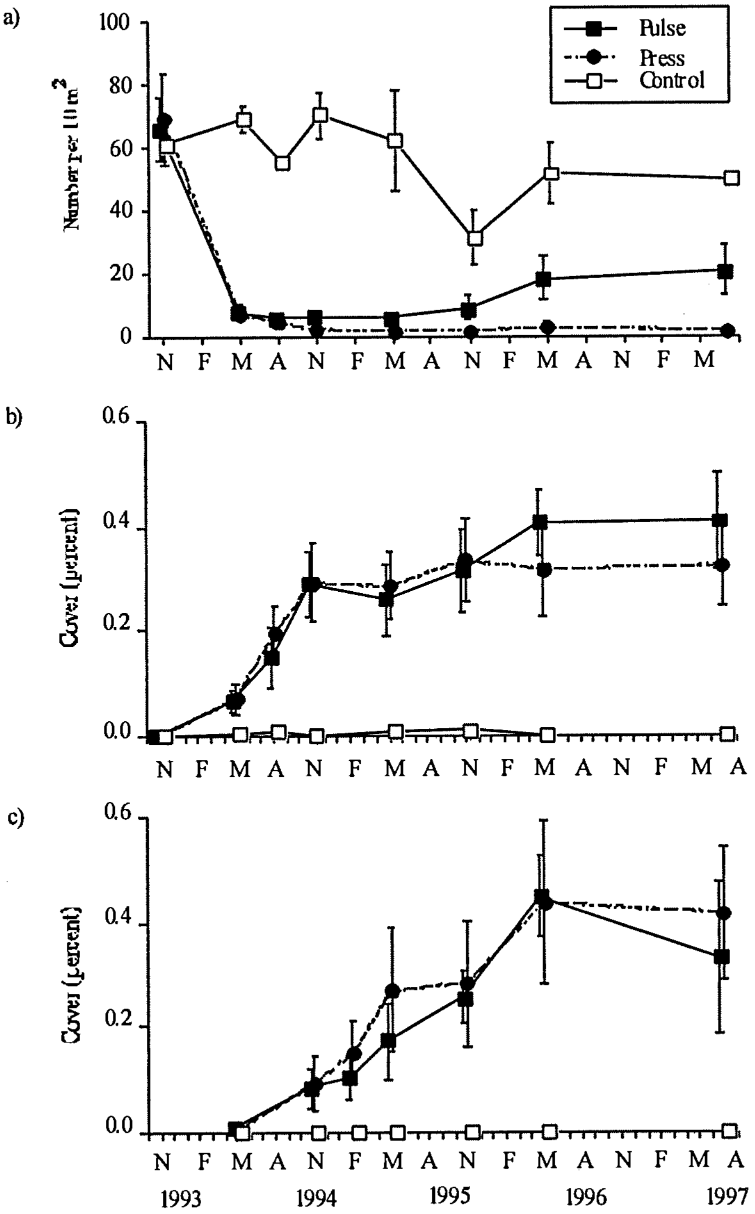


Figure 4.1 Changes through time in (a) mean density per 10 m² (+/- SE, n = 8-9) of *Centrostephanus* in the Pulse, Press and Control treatments; (b) mean percentage cover (+/- SE, n = 5-6) of secondary and tertiary algae in summer only, and (c) mean percentage cover (+/- SE, n = 5-6) secondary and tertiary algae in winter only. Symbols as indicated on the graph. Data are pooled among treatments based on analyses. Sample sizes in (a) are n = 9 replicate gutters prior to May 1995 and n = 8 between May 1995 and July 1997, and in (b) and (c) n = 6 and 5 gutters respectively.

		Urchins			Foliose Algae			*Cystophora*			Turfing Algae		
Source	df	MS	F	prob	MS	F	prob	MS	F	prob	MS	F	prob
Control v Rest	2	361.19	8.62	**	6.86	6.15	**	4.50	3.79	ns	3.37	7.73	**
Abalone	1	3.15	0.08	ns	0.01	0.01	ns	0.00	0.00	ns	0.01	0.01	ns
Urchin	1	282.61	6.75	*	0.01	0.01	ns	0.55	0.46	ns	1.94	4.45	*
Season	1	7.64	0.18	ns	0.22	0.20	ns	0.27	0.23	ns	3.25	7.46	*
Ab x Ur	1	-	-		-	-		-	-	-	-	-	
Ab x Se	1	-	-		-	-		-	-	-	-	-	
Ur x Se	1	-	-		-	-		-	-	-	-	-	
Ab x Ur x Se	1	-	-		-	-		-	-	-	-	-	
Gutter (AbUrSe)	20	41.89	39.30	**	1.12	27.13	**	1.19	25.93	**	0.44	8.23	**
Residual	270	1.07			0.04			0.05			0.23		
Transformation		sqrt			arcsin(sqrt)			arcsin(sqrt)			arcsin(sqrt)		

Table 4.2 Analyses of variance for differences in the density of urchins and percentage cover of foliose algae, *Cystophora* and turfing algae in the sea urchin removal experiment. Non-significant interaction terms were pooled following the guidelines of Winer et al. (1991). The error term used for all tests of main effects and interactions was the Gutter term. For the F tests, 6 d.f. were subtracted from the Gutter term because of 'lost' replicates (see text). Where the three factor interaction term was pooled, the degrees of freedom for tests using the Gutter term were therefore (1,15) and where all interaction terms were pooled the d.f. for tests of the main effects were (1,18). The pooled Gutter Mean Square is shown in all instances where interactions were pooled. Appropriate transformations as indicated following Cochran's test of homogeneity of variance are shown. ns indicates non-significant at $p < 0.05$, * $p < 0.05$, ** $p < 0.01$. - indicates pooled MS and no F test.

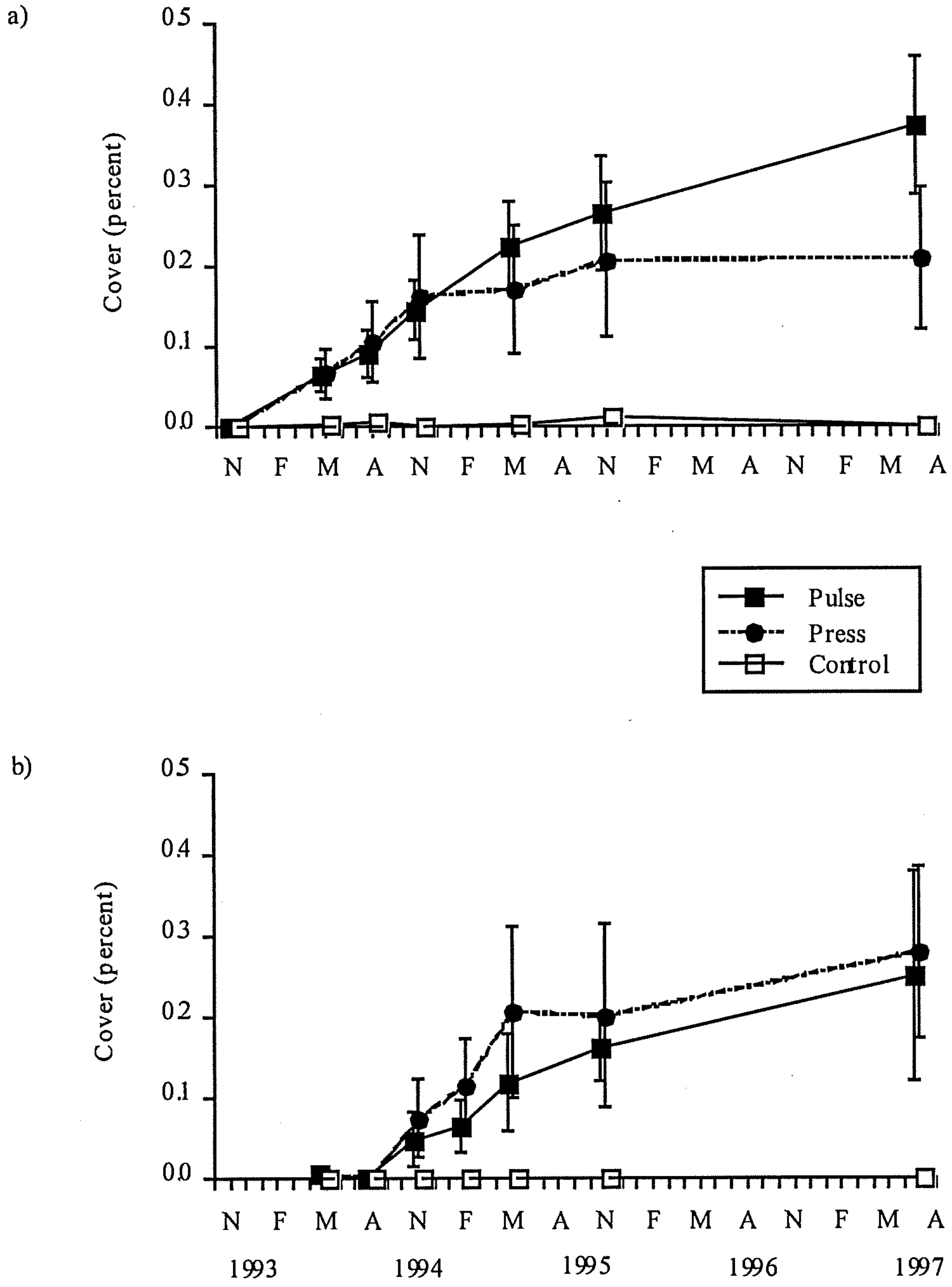


Figure 4.2 Changes in mean percentage cover (+/- SE, n = 5-6) of Cystophora through time in (a) Summer and (b) Winter. Symbols as indicated on the graph. Data are pooled across factor Abalone.

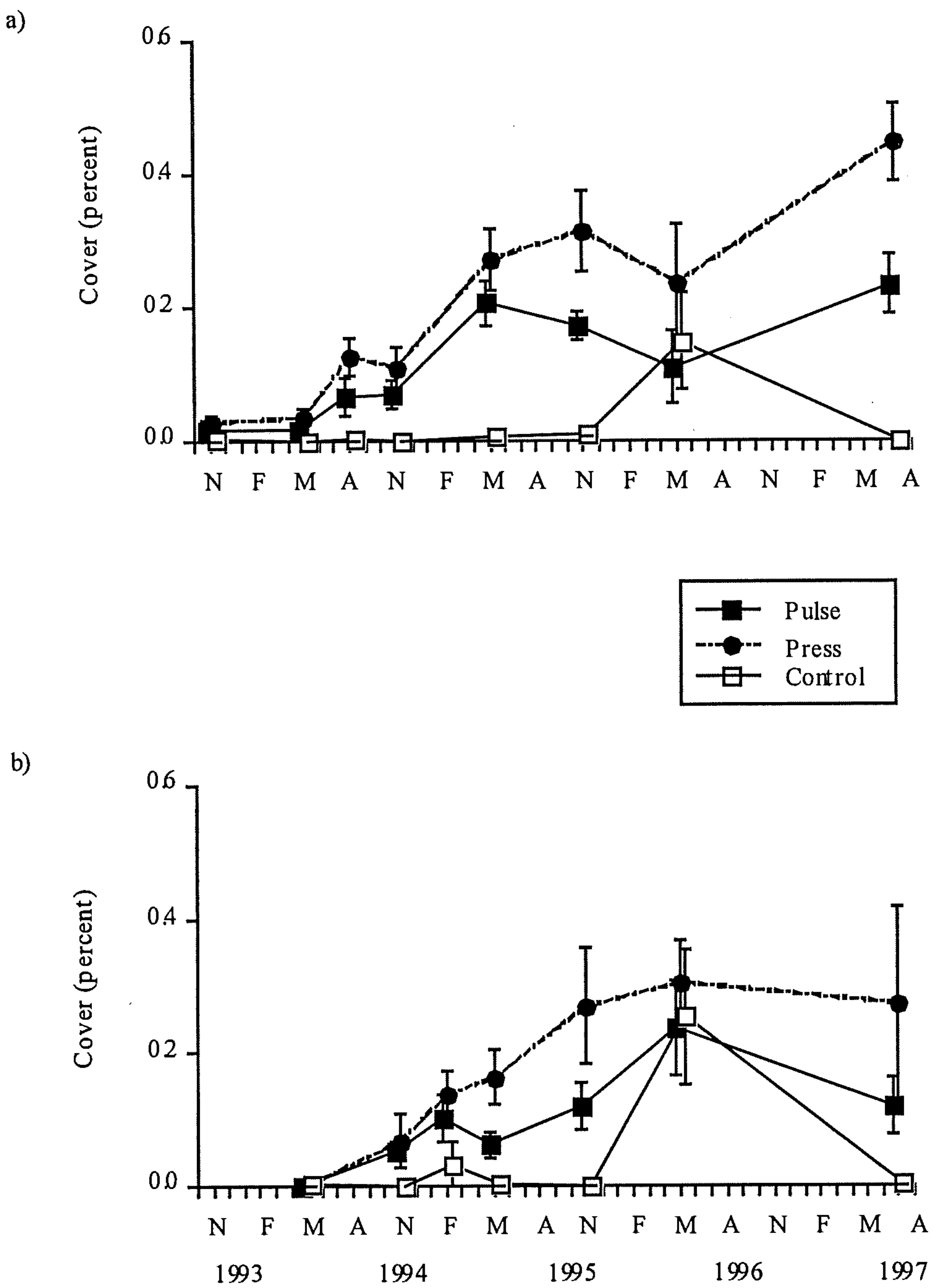


Figure 4.3 Changes in mean percentage cover (+/- SE, n = 5-6) of turfing algae through time in (a) Summer and (b) Winter. Symbols as indicated on the graph. Data are pooled across the factor Abalone.

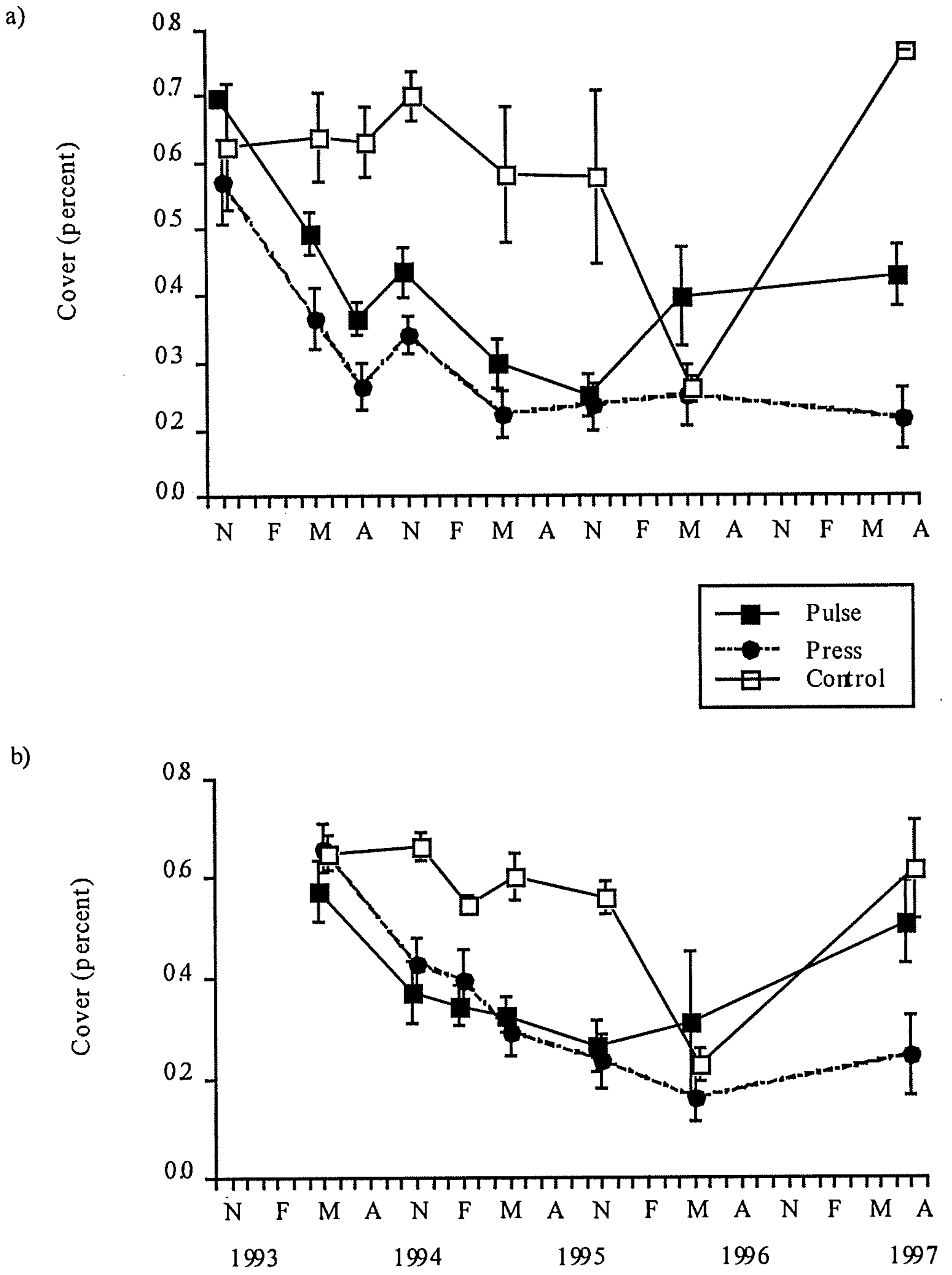


Figure 4.4 Changes in mean percentage cover (+/- SE, n = 5-6) of crustose coraline algae through time in (a) Summer and (b) Winter. Symbols as indicated on the graph. Data are pooled across the factor Abalone.

algae. In July 1997, foliose algae covered 30-40% of the secondary and tertiary 'space' in both the summer and winter treatments. Foliose algae remained all but absent from the controls over the course of the experiment (Figure 4.1b, c).

Cystophora monoliformis dominated the percentage cover of foliose algae throughout the experiment (Figure 4.2). In July 1997, *C. monoliformis* accounted for 24.5 (SE = 6.21 and 19.7 (SE = 6.88 %) of the total cover in summer and winter treatments from which sea urchins had been removed. The recruitment and growth of *C. monoliformis* was not consistent among replicate gutter. Although it dominated some gutters, it comprised only a minor portion of the macroalgae in others. These patterns are reflected in the significant differences found among gutters when analysed in July 1997 (Table 4.2). These differences were such that there were no significant effects for any of the orthogonal factors or their interactions (Table 4.2). *C. monoliformis* remained all but absent from the controls over the 3.2 years the experiment ran.

Of the remaining species of large brown algae, *Phyllospora comosa* (crayweed), *Ecklonia radiata* and *Sargassum* spp. were most abundant but the mean abundance of these species was always small (4.8 and 20.2 and 11.8 % respectively per gutter and 3.1, 7.3 and 5.1% per treatment). For all three variables there were significant differences in percentage cover among gutters in July 1997 (see Table 4.2 for model used). All three species were rare in the controls for the duration of the experiment. The effects of sea urchin removal on the abundance of *Sargassum* spp and *Ecklonia radiata* were similar to those for *Phyllospora comosa*. All three species were rare in both the summer and winter treatments until spring 1994. Large numbers of recruits appeared in August 1994 although these plants did not cover a large proportion of the substratum. The great majority of plants at this time were small (< 200 mm high).

The percentage cover of articulated corallines (turfing algae) rose in all treatment s following the removal of sea urchins (Figure 4.3). In July 1997, there were significant differences between the controls and sea urchin removal treatments (Table 4.2). The cover of turfing algae was significantly greater in the press treatment than in the pulse treatment in July 1997 (Table 4.2). The cover of turfing species was significantly greater in the Summer treatments than in those cleared six months later in Winter 1994 (Figure 4.3, Table 4.2).

The percentage cover of crustose coralline algae fell sharply following the removal of sea urchins as it was replaced as the primary cover by filamentous and turfing algae, and the recruits of foliose species. The mean percentage cover of crustose corallines in the controls remained high, except in May 1996 when the cover fell sharply because of an influx of filamentous and turfing algae (Figure 4.4). In July 1997, the mean percentage cover of crustose corallines had risen again to be greater than 60% (Figure 4.4). In the pulse treatments, the cover of crustose corallines increased slowly after November 1995 and was significantly greater than in the press treatment in July 1997 (Table 4.2). At the end of the experiment the percentage cover of crustose corallines was significantly less in the sea urchin removals than in the controls (Table 4.2). There were no significant effects of season or the addition of abalone on the percentage cover of crustose corallines, nor were there any significant interactions between these

		Small Abalone			Small Abalone			Sublegal abalone		
		July 1997			May 1995			July 1997		
Source	**df**	**MS**	**F**	**prob**	**MS**	**F**	**prob**	**MS**	**F**	**prob**
Control v Rest	2	10.93	5.02	*	12.65	6.83	*	52.19	8.83	**
Abalone	1	2.42	1.11	ns	14.88	8.03	ns	2.35	0.40	ns
Urchin	1	7.09	3.26	ns	1.22	0.66	ns	26.00	4.40	*
Season	1	2.22	1.02	ns	0.09	0.05	ns	5.74	0.97	ns
Ab x Ur	1	-	-		-	-		-	-	-
Ab x Se	1	-	-		-	-		13.75	2.33	ns
Ur x Se	1	-	-		-	-		-	-	-
Ab x Ur x Se	1	-	-		-	-		-	-	-
Gutter (AbUrSe)	20	2.18	4.95	*	1.85	2.84	ns	5.91	10.67	**
Residual	270	0.44			0.65			0.55		
Transformation		ln (x +1)			sqrt			ln (x +1)		

Table 4.3 Analyses of variance for differences in the density of small abalone in July 1997 and May 1995, and sublegal sized abalone in July 1997

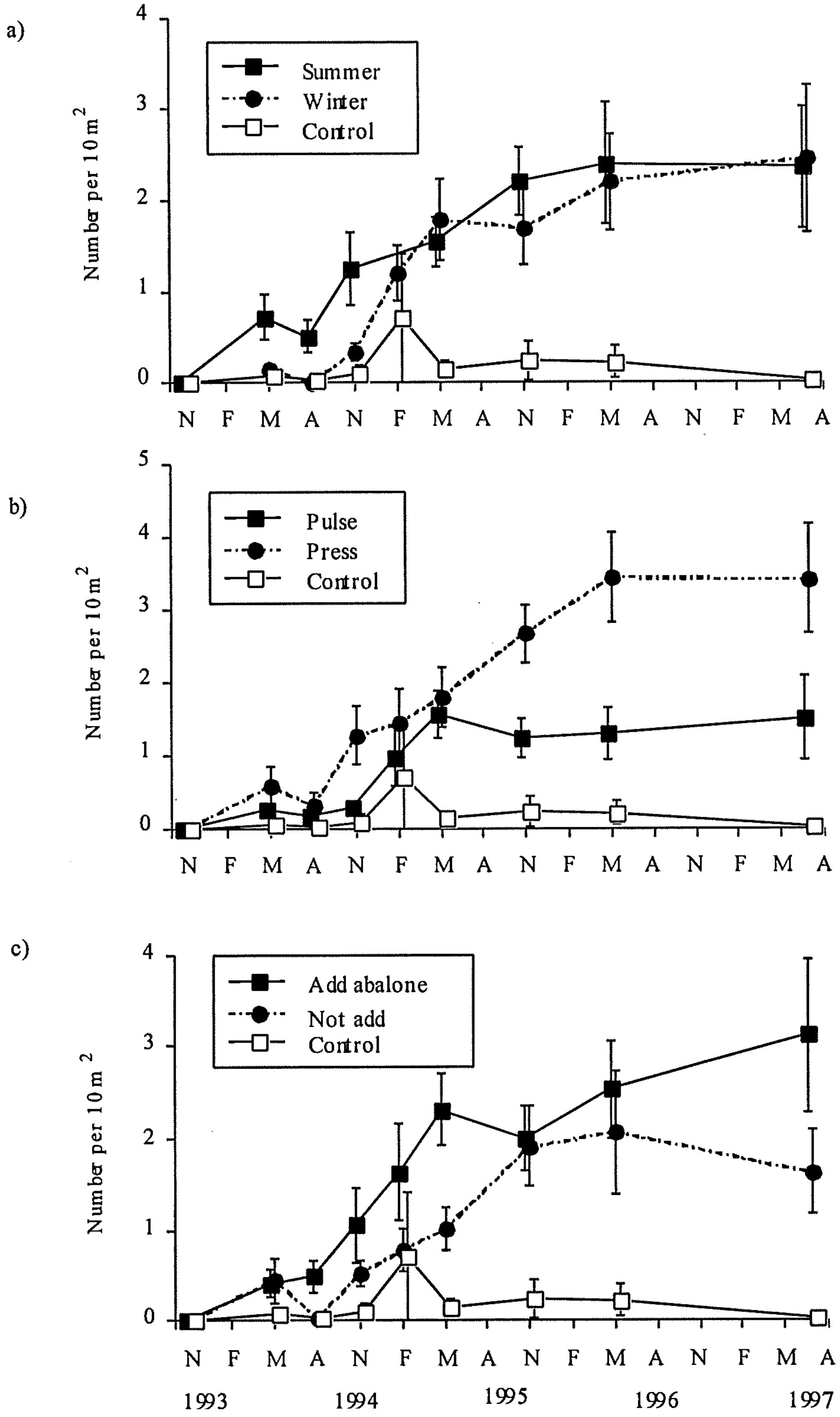


Figure 4.5 Changes in mean density of small abalone (+/- SE, n = 8-9) per 10 m2 through time in the (a) Summer and Winter treatments, data pooled across the factors Urchin and Abalone, (b) Pulse and Press treatments (factor Urchin), data pooled across the factors Season and Abalone, and (c) Add abalone and not-add Abalone treatments, data pooled across the factors Season and Urchin.

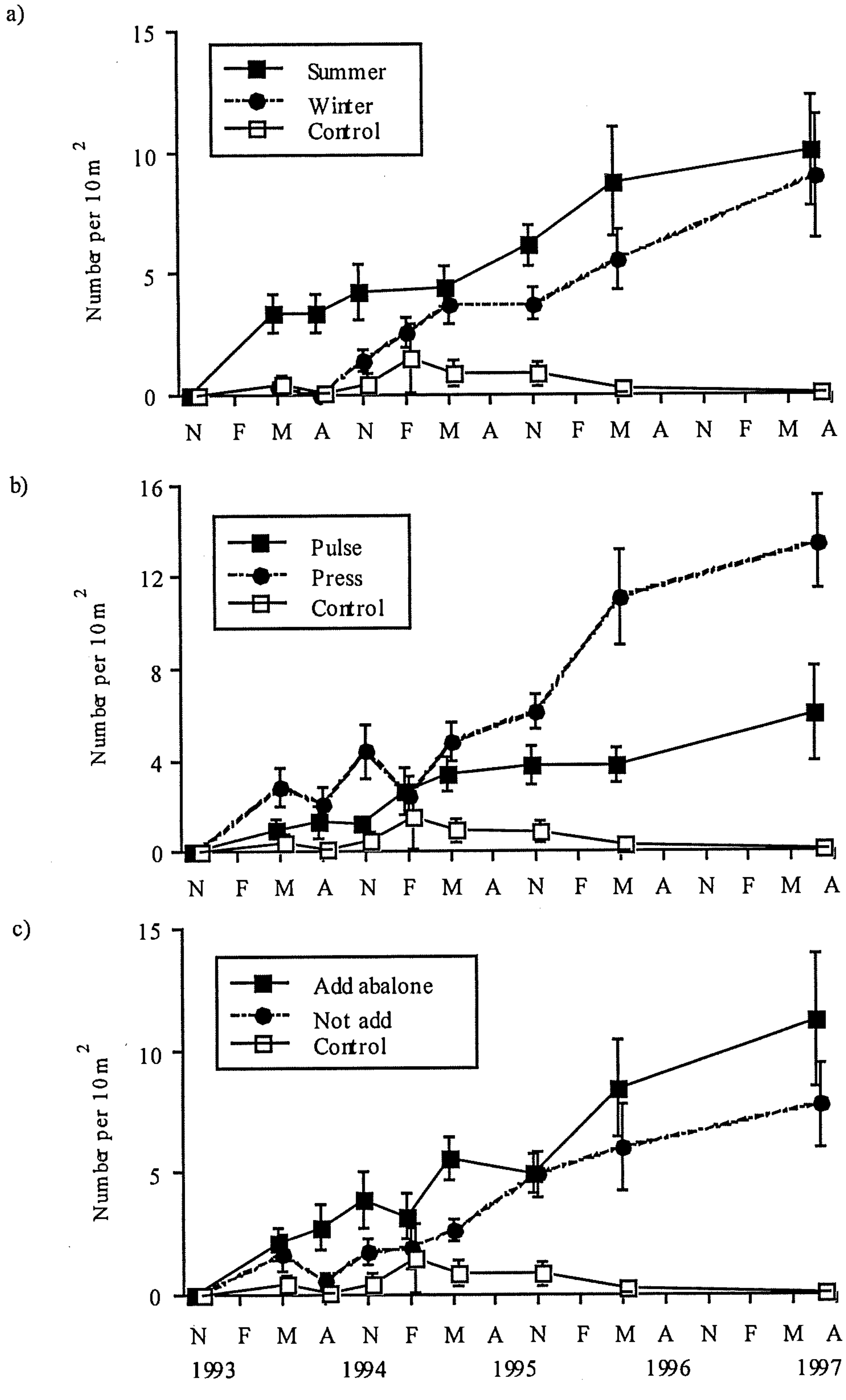


Figure 4.6 Changes in mean density of sub-legal abalone (+/- SE, n = 8-9) per 10 m2 through time in the (a) Summer and Winter treatments, data pooled across the factors Urchin and Abalone, (b) Pulse and Press treatments (factor Urchin), data pooled across the factors Season and Abalone, and (c) Add abalone and not-add Abalone treatments, data pooled across the factors Season and Urchin.

factors (Table 4.2).

Abalone

Small abalone (<65 mm) were rare in the gutters prior to the removal of sea urchins (Figure 4.5a). Densities in the control gutters remained less than 0.2 m^{-2} throughout the experiment, except in February 1995 when an influx of small abalone were recorded in one replicate only (note large variance associated with this mean, Figure 4.5a). When analysed for differences among treatments, in July 1997, there were no significant interactions among factors. These terms were therefore pooled and the results are presented separately for each main effect (Figure 4.5).

The removal of *Centrostephanus* had a large impact on the density of small abalone (Figure 4.5). The rate of increase in density slowed after May 1995 however, there were still significant differences between the controls and all treatments from which *Centrostephanus* had been removed at the end of the experiment (Table 4.3). The combined mean density of these latter treatments in July 1997 was 2.4 (SE = 0.51) compared to 0.02 (SE = 0.02) in the controls. At the end of the experiment there were no significant effects of the timing of urchin removal, the addition of broodstock, or the maintenance of the removals, nor did these factors interact (Figure 4.5, Table 4.3).

When analysed in May 1995, there were significantly more small abalone in those gutters in which broodstock had been added (Figure 4.5a, Table 4.3), in fact the addition of broodstock doubled the effect of removing sea urchins (Figure 4.5a). At that time, neither Season or the type of urchin removal, nor their interactions significantly influenced the density of small abalone (Table 4.3). In May 1995, in those gutters from which sea urchins had been removed but with no broodstock, the densities of small abalone had increased by an order of magnitude by May 1995 (Figure 3.2.1b). In the summer removals, done in November-December 1993, densities initially rose but declined in early spring 1994, before rising again during early summer of 1994. In the winter removals, densities rose over the late summer of 1994-95 and remained at high levels in May 1995 (Figure 4.5). In May 1995 there was no significant difference in the density of small abalone between the summer and winter removals, despite the difference in time elapsed since the removals (Table 4.3).

Small abalone were defined as being smaller than 65 mm. Growth estimates (Worthington et al. 1995, Andrew et al. 1997, Worthington and Andrew 1998) indicate that, over the course of the experiment, small individuals would grow out of this size category but would still be less than 115 mm in length at the end of the experiment. For this reason, the combined densities of all abalone smaller than 115 mm were analysed. The analysis and interpretation of differences among treatments in the abundance of this class of sizes would not be confounded by the introduction of broodstock in November 1993 and May 1994 because the great majority of these animals, all of which were > 90 mm when introduced would have grown to be greater than 115 mm in the intervening 3.2+ years (Andrew et al. 1997).

In July 1997, the densities of sub-legal sized abalone in the gutters without *Centrostephanus* had risen to an average of 9.6 (SE = 1.66, pooled across all treatments). In the controls, the densities remained small (Figure 4.6) and were 0.12 (SE = 0.06) at the end of the experiment. When analysed this effect was significant, as was the difference between the pulse and press treatments (Figure 4.6b, Table 4.3). There were significantly more sub-legal abalone in those gutters in which the sea urchin removals were maintained (Figure 4.6b). There were no significant differences attributable to the introduction of broodstock or the timing of removal, nor did these factors interact (Table 4.3). There were significant differences among gutters in the density of sub-legal abalone (Table 4.3).

4.2.3 Extension of Results to Industry

Commercial abalone divers were kept informed of the developing results from the small scale experiment in Eden (described above) both through regular updates at Industry Open Days, letters to divers and through ABMAC meetings. As part of this process, divers identified sites from they had abalone to be declining and urchins increasing in abundance. Among these sites nominated by divers were those the believed had been 'lost' to the fishery because of the increasing densities of sea urchins. In response, large scale experiments were designed to test the generality of the results from Twofold Bay and to 'scale up' the removals.

With the assistance of commercial divers, urchins were cleared from larger areas of reef (> 1 hectare) between Port Stephens and South Saltwater Beach, south of Eden (Table 4.4). These sites have been maintained at irregular intervals since and form the basis of an ongoing study of the ecological impact of the developing sea urchin fishery and its relationship with the abalone fishery. The results of these 'experiments' will be reported separately.

Table 4.4. Summary of sites from which *Centrostephanus* were removed and the dates of the initial removals.

Port Stephens	18-20 February 1997 and 11 June 1997
Curracorang	18-26 March and 27-28 May 1997
Brush Island	22-23 January, 10-13 February, 24-26 February, 3-5 March and 8 May 1997
Jimmy's Island	17 November 1993, 8 March 1994, 8 February 1996, 6 March 1997
Eden	17 April 1994 and 8 July 1997
Mowarry Rock	8-9 April and 20 April 1997
South Saltwater Beach	10 April and 22 May 1997

4.2.4 Conclusions

Although the importance of interactions among fisheries is increasingly appreciated, rarely is the link so clear and its study so tractable as it was in the present case. Sea urchins and abalone in NSW live on shallow subtidal reefs, may be directly observed, move relatively small distances, and may be experimentally manipulated. As such they offered an important opportunity to understand the ecological relationship between two exploited species. This understanding has facilitated developing management strategies for the abalone fishery and the developing sea urchin fishery.

Our experimental manipulation of sea urchins and abalone suggests the following:

1. Removal of sea urchins causes a change in habitat, from the Barrens habitat to one dominated by foliose algae. The species composition of colonising algae was not predictable. In some gutters articulated corallines dominated whereas in others, different species of large brown algae were most abundant. There were no clear effects of timing of urchin removal on the species composition of cleared areas.

2. Removing sea urchins from gutters caused the densities of abalone to increase by an order of magnitude.

3. The effect of adding broodstock abalone on the densities of small abalone was evident up to 12 months after introduction but disappeared thereafter.

These results suggest that, given the predominance of the Barrens habitat on reefs throughout the range of the NSW abalone fishery (Chapter 3), sea urchins may limit the productivity of the New South Wales abalone fishery. The development of a sea urchin roe fishery offers the potential to enhance the abalone fishery by reducing the abundance of sea urchins on reefs. Although such a fishery is likely to concentrate on sea urchins from the Fringe habitat in the first instance, the expansion of the fishery into other habitats may follow. This expansion is likely to come about through more sophisticated management such as rotational closures, and possibly the enhancement of roe quality through control of urchin densities and post-harvest processing (Chapter 5).

The significant increase in the density of small abalone one year after the addition of broodstock is intriguing. If this result is 'real', then it provides experimental support for previous work suggesting that blacklip abalone may have a measurable stock - recruitment relationship over small spatial scales (e.g. Prince et al. 1988a, b, McShane 1995, Shepherd and Partington 1995). Adjacent gutters used in this experiment were separated by distances of less than 50 m. The clear differences among gutters with and without broodstock over this scale and over this timeframe suggests that management imposed at small scales may be effective. Further, it suggests that the localised depletion of abalone from reefs, caused by poaching or freshwater run-off, is likely to have a lasting impact on the local abundance of abalone.

The disappearance of the effect of adding broodstock in year two to five of the experiment may have been caused by many factors. In the last 3 years of the experiment, it is possible that abalone recruited into the experimental gutters had become reproductively mature and were contributing to local settlement and recruitment. More probably, there is likely to be enormous variation in the scale of dispersal of abalone larvae. Sasaki and Shepherd (1995) have demonstrated such variability in the dispersal *of Haliotis discus hannai*. Dispersal will be the product of a wide range of influences, including sea surface conditions, currents, and bottom topography. These factors may well have contributed to the results obtained in the first year - either in concert with or instead of the localised effect of adding broodstock.

As a consequence of this finding, larger scale 'experiments' to test the hypothesis of small scale (in space and time) stock-recruitment relationships have been instigated. At the next scale up, the large-scale urchin removals described above will promote the development of dense populations of abalone in areas of about 1 hectare or larger. The relative recruitment strength of abalone into these areas, although confounded by changes in habitat will provide auxiliary support or counter evidence for the hypothesis of localised stock-recruitment relationships in abalone. Similarly the sea urchin fishery will provide interesting comparisons by causing large changes in the abundance of *Centrostephanus* in some subzones. A more interesting test of this hypothesis, and one not confounded by changes habitat, will come through a large-scale manipulation of abalone density within the abalone fishery. In 1998, the Industry has agreed to closures of sub-zones to fishing during the spring spawning season. Increased densities of small abalone in these closed subzones relative to fished controls would provide strong evidence for a stock-recruitment relationship at the kilometre scale. Effects will be quantified through the analysis of catch and effort statistics and fishery-independent sampling. To our knowledge this is the first such 'experiment' to be conducted in Australia and, if successful will provide a strong impetus for management on a smaller spatial scale than is currently the case. For example, within the present TACC structure, rotational closure of subsets of the 83 subzones in the fishery may the basis for enhancing the fishery. Given the high growth rates recorded for this species, there is likely to be both yield and recruitment gains from such closures. Consideration should also be given to longer term closures of subzones such as those presently in place in Sydney (e.g. Caddy 1989, Sims 1992, Quinn et al. 1993, Rogers-Bennett et al. 1995, Holland and Brazee 1996).

We conclude that it is possible to enhance the densities of abalone in areas of reef in NSW by removing *Centrostephanus rodgersii*. The mechanism underlying this effect is unclear because removing this sea urchin causes large changes in habitat structure as well as increasing available shelter. The implications of these changes to the development of a sea urchin roe fishery are discussed separately (see Chapter 5).

5. Factors Affecting Roe Quality in *Centrostephanus rodgersii*

Objective 4. To describe the effects of habitat type, sea urchin density and harvest time on the quality of roe harvested from sea urchins and to gather preliminary data on the potential for a sustained commercial fishery for this species in New South Wales.

The greatest yields of high quality roe from *Centrostephanus* were obtained from the Fringe habitat in summer and autumn. Although post-harvest treatment of the roe may enhance roe quality to some extent, it is probable that roe harvested from the Barrens habitat at any time and from algal-dominated habitats at other times will be either of poor quality and/or have an uneconomically low recovery rate. We recommend fishing practices that promote high yield of quality roe by paying divers for the weight or quality of roe harvested rather than the weight of sea urchins landed. Paying by weight of sea urchin will promote larger harvests of poorer quality urchins. Management of the roe fishery using small-scale zones within the abalone fishery will further promote high yields by reducing sea urchin density and allow the ecological impacts of the sea urchin fishery to be quantified.

5.1 Preamble

The work presented in this chapter is an abridged version of a paper published as:

M. Byrne, N.L. Andrew, D.G. Worthington and P.A. Brett (1998). Reproduction in the diadematoid sea urchin *Centrostephanus rodgersii* in contrasting habitats along the coast of New South Wales, Australia. Marine Biology. In Press.

5.2 Introduction

There has been an enormous increase in the commercial exploitation of sea urchins in recent years. The history of several major urchin fisheries has, however, been one of explosive growth then overfishing (e.g. Kalvass 1992, Vadas and Steneck 1995, Pfister and Bradbury 1996). In Japan, sea urchin production is largely underpinned by enhancement. To our knowledge there are no commercial fisheries for temperate sea urchins in the family Diadematoida. In southeastern Australia, *Centrostephanus rodgersii* is abundant and grazing by this species plays a major role in structuring benthic communities (Fletcher 1987, Andrew and Underwood 1989, 1993, Andrew 1991, 1993). *Centrostephanus* is found from the subtropical region of northern New South Wales (28°37'S; 153°38'E) to the cool temperate

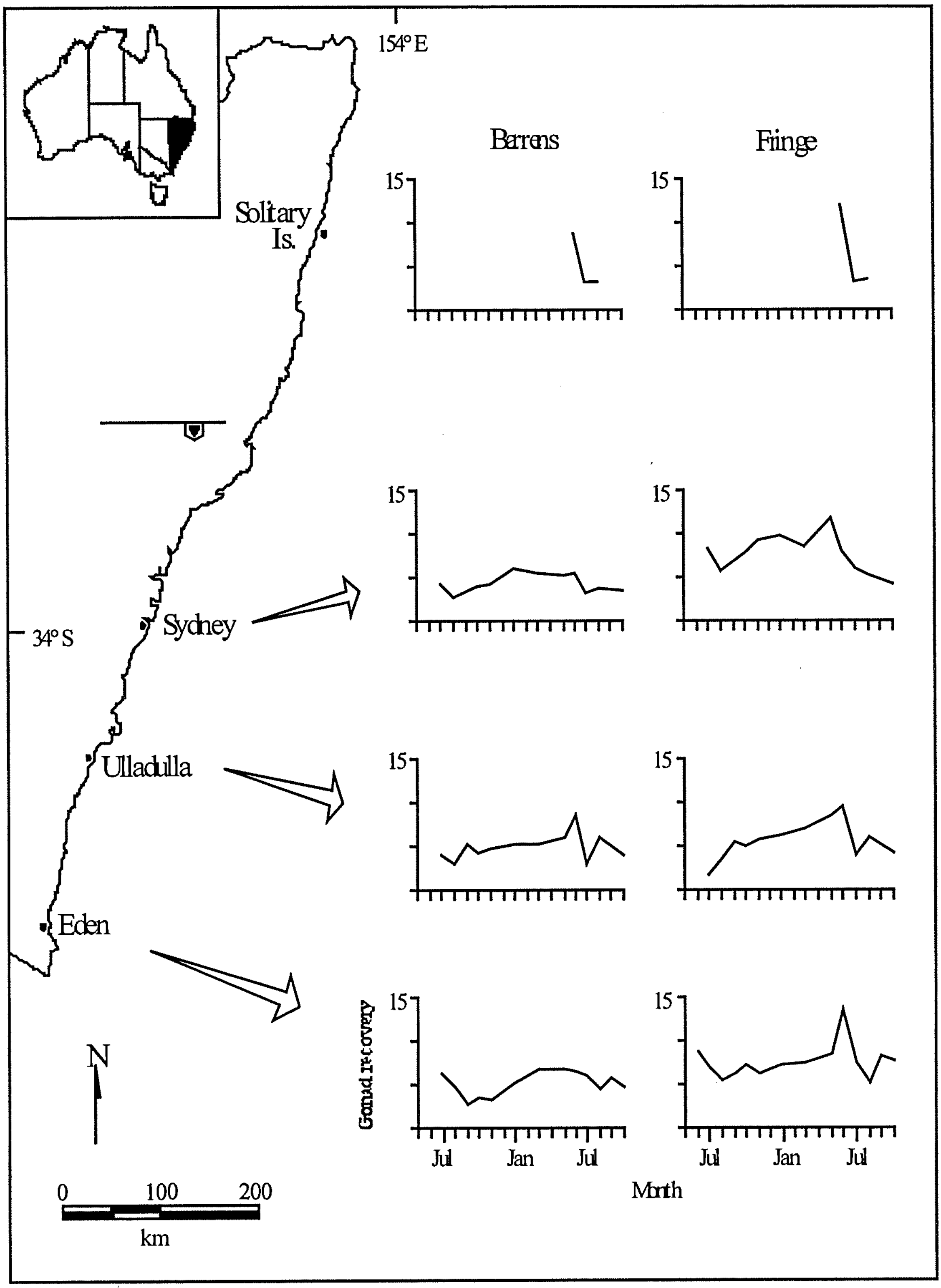


Figure 5.1 Sampling locations along the coast of New South Wales and trends in gonad recovery rate recorded at each site.

regions in eastern Victoria and northern Tasmania (41°52'S; 148°18'E). Investigations of reproduction in *Centrostephanus* indicate that this species has a short, intensive breeding season near its northern limit and prolonged breeding in the middle of its range (O'Connor et al. 1978, King et al. 1994). In the present study, the influence of photoperiod and temperature on reproduction is examined in the contrasting environmental conditions found in the subtropical and temperate waters of New South Wales.

At each location we quantified differences in reproduction between contrasting Barrens and Fringe habitat types. In New South Wales the Barrens habitat is found in areas subject to intense grazing by *Centrostephanus rodgersii.* This habitat is defined in part by the absence of macroalgae and the high crustose coralline algal cover typical of urchin barren grounds (Andrew and Underwood 1989, 1993, Andrew 1991, 1993, Underwood et al. 1991). In contrast, the Fringe habitat has a high biomass of macroalgae and, although *Centrostephanus* may be present, they are not sufficiently abundant to remove all the macroalgae and create a Barrens habitat (Andrew and Underwood 1989, 1993, Andrew 1991, Underwood et al. 1991). No attempt was made to quantify the biomass of algae in the Fringe habitats sampled. The comparisons made are therefore between areas with no macroalgae (Barrens) and areas with large but variable standing crops of laminarian and fucoid algae (Fringe). The gonads are the main nutrient storage organ in sea urchins and in food-poor Barrens conditions a lower gonad yield would be expected (Pearse 1981, Lawrence and Lane 1982, Andrew 1986, Byrne 1990).

A developing commercial fishery for *Centrostephanus* exists in New South Wales. Previous studies in the Sydney region indicate that a spring/summer harvest is required to obtain roe with the firm texture preferred by the market (King et al. 1994). This texture is characteristic of gonads dominated by nutritive cells rather than gametes. We assessed the generality of this result at major fishing ports spread across a wide geographic range to provide advice on times of year in which the yield and quality of roe per urchin can be maximized.

5.3 Materials and Methods

Reproduction of *Centrostephanus rodgersii* in Barrens and Fringe habitats was investigated at Sydney (33°54'S 151°17'E), Ulladulla (35°23'S 150°29'E) and Eden (37°06'S 149°55'E) (Figure 5.1). Collections from Eden and Sydney started in January and February 1994, respectively and those from Ulladulla started in July 1994. Twenty to thirty urchins (66-120 mm test diameter) were collected at approximately bimonthly intervals from each location until May 1995 and thereafter at monthly intervals from June to October 1995. *Centrostephanus* was also collected from Barrens and Fringe habitats from May to August 1995 and in June 1997 at the Solitary Islands (30°08'S 153°14'E) near the northern limit of its range.

Each urchin was weighed and the weight of gonad per urchin was estimated by either weighing the whole gonad or scaling up the weight of one randomly selected undamaged element. Although the gonad index (gonad weight as a fraction of body weight) is typically used in studies of urchin reproduction, it is inappropriate for studies such as this where a wide range of sizes of urchins are sampled (Gonor 1972, Grant and Taylor 1986). We used the gonad recovery rate (GRR) method which is calculated as the slope of a regression of gonad weight against total weight. Regressions were fitted separately for each sample using a common non-zero intercept. When different intercepts were fitted among samples, they were not significantly different.

For each location, the maximum GRR recorded for the Barrens and Fringe habitats in 1994-95 were compared by a two factor orthogonal analysis of variance. The factors analysed were Location (Solitary Is., Sydney, Ulladulla, Eden) and Habitat (Fringe, Barrens). For this analysis, twenty urchins were randomly selected from each sample. For the sample of urchins from the Solitary Is. collected in June 1997, a t-test was used to compare the urchins from Fringe and Barrens habitats. The influence of test diameter on GRR was examined by linear regression.

For histology, one gonad from each urchin was fixed in Bouin's fluid. The gonads were dehydrated in ethanol, embedded in paraffin and sectioned. Sections (7 µm thick) were stained with haematoxylin and eosin. The gametogenic condition was categorized into five maturity stages based on the morphological criteria used in King et al. (1994). The criteria for four of the stages: spent, recovery, growing and partly spawned stages are as detailed in King et al. (1994). The mature stage used here combines the premature and mature stages in King et al. (1994). Although these broad categories assist in presenting the overall pattern of gonad development, among-location differences in the details of gametogenesis during the breeding season were not well presented by this method. For the among-location comparison during breeding, the gametogenic status of the ovaries was examined in detail and recorded photographically.

Sea surface temperatures for the four study locations were obtained from the CSIRO. Temperature was estimated by satellite using a grid of nine points around the coordinates for each site. Mean monthly temperature was calculated from mean temperature across 10 days which was recorded every 5 days. Times of sunrise and sunset, obtained from the Sydney Observatory, were used to estimate day length.

5.4 Results

Gonad growth and gametogenesis

Seasonal patterns in the GRR and gametogenesis in *Centrostephanus rodgersii* were similar at the four locations (Figures 5.1-3). Across all locations the reproductive cycle was divided into four main phases which were most easily discerned in the Fringe habitat. In both males and females these phases were: 1) a major spawning period in winter starting in June; 2) a period of post-spawning recovery; 3) a period of gonad weight increase between August and April due to enlargement of the nutritive phagocytes and; 4) a period of intense gametogenesis in May when the gonads contain rapidly developing gametes in

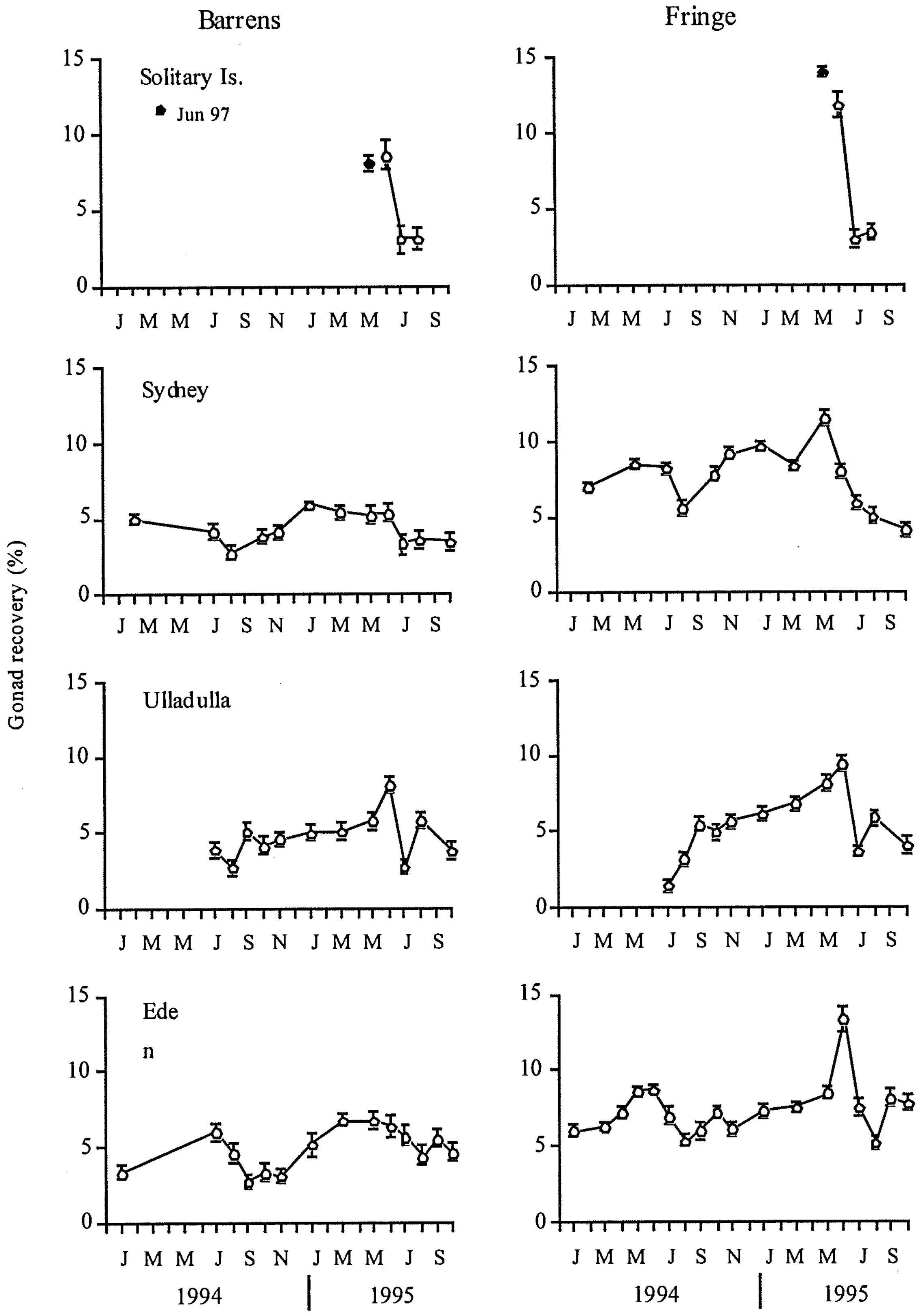


Figure 5.2 Gonad recovery rate (mean + SE, n= 20-30) for Centrostephanus at the Solitary Islands, Sydney, Ulladulla and Eden. The GRR recorded for the Solitary Islands in June 1997 is indicated.

preparation for spawning (Figure 5.3). Within these broad phases there were differences in the gametogenic state of the gonads at the four locations as detailed below (Figures 5.3). During the breeding season these differences were most apparent in the ovaries.

Solitary Islands

Gonad development of *Centrostephanus rodgersii* at the Solitary Islands was highly synchronous with little or no variability in gametogenic condition among urchins and habitats (Figure 5.3). All urchins collected on 11 May 1995 had growing gonads at an early stage of gametogenesis. The gonads were strongly eosinophilic due to the abundance of nutritive phagocytes which filled the lumen. The ovaries were lined with pre and early vitellogenic oocytes and the testes had a layer of developing sperm in short spermatocyte columns. When the next sample was collected on 27 June, spawning had begun. The ovaries contained an abundance of ova (100-110 μm diameter) and the testes had a large store of spermatozoa. The period between 11 May and 27 June was characterised by synchronous and rapid gametogenesis. The 10 June 1997 sample illustrated the gametogenic condition and GRR just prior to spawning (Figure 5.2). At this time the females contained late vitellogenic oocytes and a few ova.

Synchronous development of the entire annual cohort of gametes resulted in exhaustion of the nutritive tissue. Vitellogenesis was completed within one month. The absence of developing gametes on 27 June indicated that gametogenesis was completed for the 1995 breeding. By July the GRR was at its lowest and the gonad condition indicated that the urchins had finished spawning for some time (Figure 5.2). The absence of gametes in most urchins collected in July and August made it difficult to identify the sex of the urchins.

In 1995, spawning at the Solitary Islands occurred between 11 May and 27 June (Figure 5.2) in what appeared to be a complete and short-lived event. The mean GRR recorded in June 1995 for the Fringe and Barrens, 12.0% and 8.8%, respectively (Table 5.1), underestimated the maximum gonad output at the Solitary Islands because spawning had started. Gonad weights were not recorded for the pre-spawning May 1995 sample. The 1997 sample indicated that the gonads were in a pre-spawning condition on 10 June; the GRR measured on that date for the Fringe and Barrens habitats were 14.0% and 8.0% respectively (Table 5.1). These indices provide an estimate of the maximum gonad output for *Centrostephanus* at the Solitary Islands in 1997.

Sydney

There was little change through time in the GRR in the Sydney Barrens habitat except for a slight rise between the end of spawning in August 1994 and the beginning of spawning in June 1995 (Figure 5.2). This was followed by a slight decrease in the GRR due to spawning in July. In contrast, gonad growth of the urchins in the Fringe habitat exhibited greater seasonal change resulting in a pre-spawning peak in the

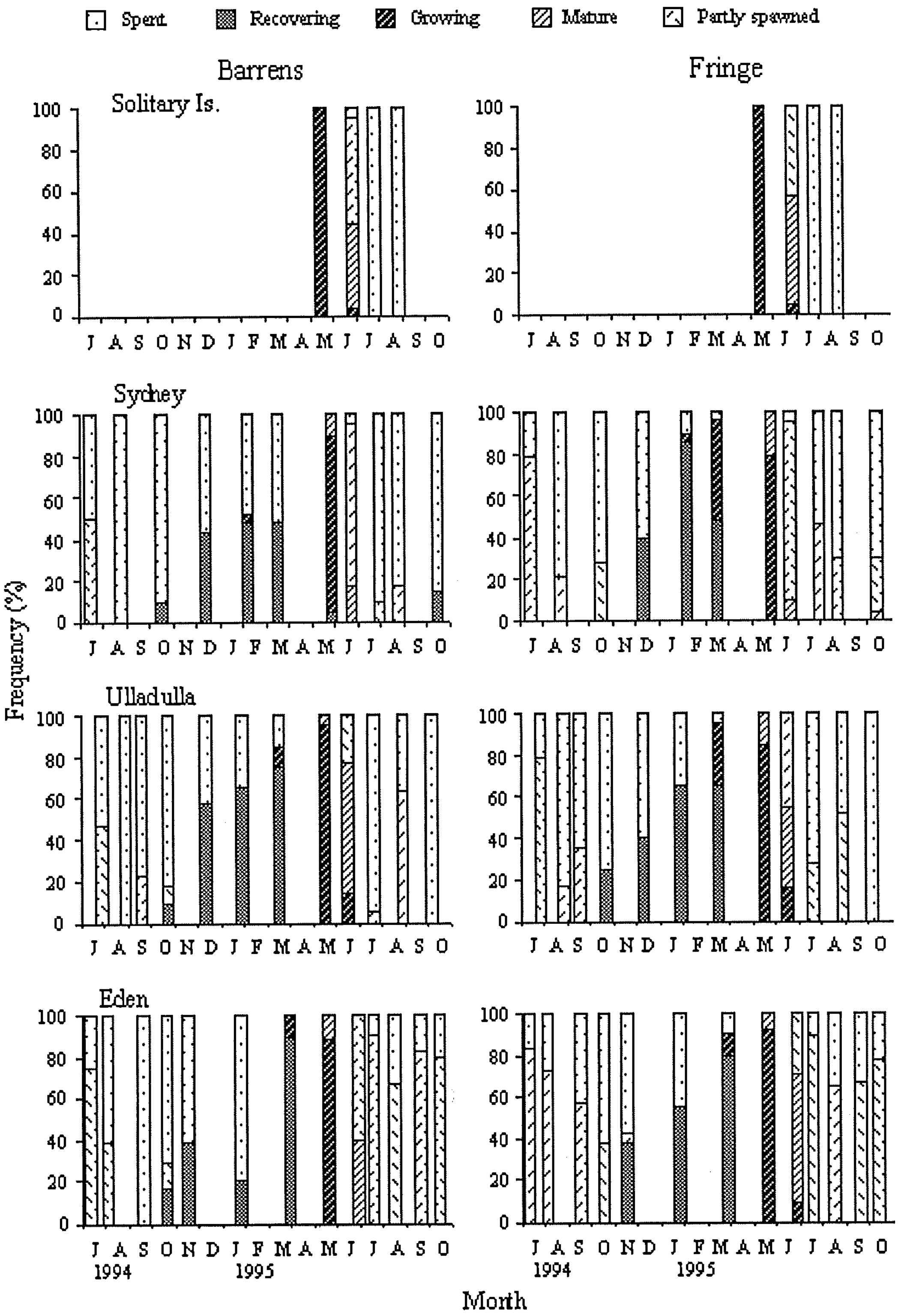


Figure 5.3 Gametogenic cycle of *Centrostephanus*. Histograms show relative frequencies of gonad stages in male and female urchins from July 1994 to October 1995 at Sydney, Ulladulla and Eden and at the Solitary Islands May to August 1995 (n = 20-30).

GRR in May followed by a sharp decrease due to spawning (Figure 5.1,2).

Despite the difference in the GRR between the Fringe and Barrens sites, histology revealed that gonad condition was similar at both sites (Figure 5.3). Gametogenic synchrony was most evident just prior to and during the early part of breeding. All urchins collected on 26 May 1995 had growing or mature gonads and the nutritive tissue was reduced (Figure 5.3). The ovaries contained an abundance of mid and late vitellogenic oocytes which were beginning to take up a central position and ova were present. Vitellogenic oocytes continued to develop in the germinal layer. The testes had well-developed spermatocyte columns giving rise to spermatozoa. On 22 June the gonads were filled with mature gametes and spawning had begun (Figure 5.3). In some gonads the germinal layer contained developing gametes which would be spawned later in the season. In July, the gonads of urchins from the Fringe and Barrens sites differed markedly (Figure 5.3). By the end of July most urchins from the Barrens habitat had finished spawning and it was difficult to discern their sex. In contrast, those in the Fringe habitat had a considerable store of mature gametes present and this population continued to spawn through October (Figure 5.3). The germinal layer of the Fringe urchins was inactive from July onwards indicating that gametogenesis was finished for the season.

Data from histology indicated that spawning by *Centrostephanus rodgersii* in Sydney started in mid to late June (Figure 5.2). During the first weeks of the breeding season spawned gametes were replaced in the lumen due to ongoing gametogenesis. The GRR dropped sharply between June and July. This was due to a complete spawn-out at the Barrens site, possibly limited to a single episode. In contrast, urchins at the Fringe habitat spawned through October in both years due to prolonged gamete storage and episodic release. The Barrens urchins spawned for approximately two months, while those at the Fringe site spawned for approximately four months. The maximum GRR recorded for the Sydney Fringe and Barrens sites were, 11.6% and 6.0%, respectively (Table 5.1).

Ulladulla

The pattern of gonad growth at Ulladulla was similar in the Fringe and Barrens sites (Figure 5.2). After the 1994 spawning period, the GRR increased from August 1994 to a maximum in June 1995, followed by a sharp drop due to spawning. All urchins collected from Ulladulla on 16 May 1995 were in pre-spawning condition with gonads at an advanced stage of gametogenesis (Figure 5.3). The ovaries contained mid and late vitellogenic oocytes and a few ova were present. The testes contained well-developed spermatocyte columns and were accumulating spermatozoa. This gametogenic activity was accompanied by a reduction in the nutritive tissue. On 14 June the gonads were filled with mature gametes and some urchins had spawned (Figure 5.3). The germinal layer contained late stage developing gametes.

In July 1994 and 1995 *Centrostephanus rodgersii* from the Barrens and Fringe sites contained partially spawned or spent gonads (Figure 5.3). The germinal layer did not contain developing gametes indicating that gametogenesis was finished for the season. As a result, subsequent spawning was due to storage of mature gametes. In August 1994 most of the urchins were spent, whereas in 1995 gametes were still present in the gonads at both sites. The rise in the GRR between July and August (Figure 5.2) was not due to renewed gametogenesis and may reflect an increase in nutritive tissue. The final decline in the GRR between August and October was due to final gamete release. By October all the urchins were spent (Figure 5.3).

In 1995, *Centrostephanus rodgersii* from Ulladulla initiated spawning between the May and June sampling dates with major activity between June and July and a final period of gamete release between August and September. Urchins from both the Fringe and Barrens sites had a four month breeding period. The maximum GRR recorded for these sites were, 9.5% and 8.3%, respectively (Table 5.1).

Eden

At Eden, the pattern of gonad growth was similar at the Fringe and Barrens sites (Figure 5.2). The cycle of change in the GRR was, however, less pronounced for the Barrens urchins. From the end of breeding in November 1994 to March 1995 the GRR increased due to expansion of the nutritive phagocytes (Figures 5.2, 5.3). The GRR was greatest in late June 1994 and 1995 followed by a decrease due to spawning (Figure 5.2).

All the urchins collected from Eden on 18 May 1995 had growing or mature gonads and most had a substantial store of nutritive tissue (Figure 5.3). The ovaries contained mid and late vitellogenic oocytes, but ova were not present. The testes had well-developed spermatocyte columns and were accumulating spermatozoa. By 28 June the gonads were packed with mature gametes and spawning had started (Figure 5.3). Mid and late vitellogenic oocytes continued to develop in preparation for release and some nutritive tissue remained at the periphery. In July and August 1994 and 1995 most urchins had partly spawned gonads. During these months a conspicuous store of gametes was present and the germinal layer was largely inactive. Stored gametes were present in the gonads of the Fringe urchins through October of both years and at the Barrens in 1995. In both years there was little change in gonad condition from July to October at the Fringe site. The rise in GRR between August and September may reflect an increase in nutritive tissue (Figure 5.2).

In 1994 and 1995 gamete release by *Centrostephanus rodgersii* at Eden started between the May and June sampling dates with the major period of gamete release between late June and August. This was followed by prolonged gamete storage and episodic spawning through October at both sites in 1995, while in 1994 spawning at the Barrens site was finished by September. At Eden *Centrostephanus* had a five to six month breeding season between June and October. The maximum GRR recorded for the Eden Fringe and Barrens sites were 13.5% and 6. 9%, respectively (Table 5.1).

Influence of habitat on reproductive output and breeding

Although the timing of gametogenesis was similar in the Fringe and Barrens sites at all locations, reproductive output differed markedly between locations and habitats (Table 5.1) . The effect of habitat was dependent on location (significant Habitat x Location effect, $F_{(3,152)} = 8.03$, $p < 0.001$). Analysis of the GRR data recorded just prior to initiation of spawning at Sydney and Eden revealed that the Fringe urchins had significantly larger gonads compared with their Barrens conspecifics (*a posteriori* Ryan's test). At Ulladulla the pre-spawning GRR did not differ between the Fringe and Barrens sites (Ryan's test). A similar result was obtained for the Solitary Islands on 27 June 1995. Spawning however, had already begun. Comparison of the pre-spawning GRR obtained for the Solitaries on 10 June 1997 provided a better indication of the maximum gonad output at this location and showed that the Fringe urchins had significantly larger gonads than Barrens urchins ($t_{(48)} = 6.74$, $p < 0.001$) at that time.

At Sydney and Eden the reproductive output of urchins in the Barrens habitat was particularly low, the maximum GRR was approximately half that recorded for the Fringe urchins (Table 5.1). The smaller gonads of Barrens urchins correlated with their shorter breeding period compared with the Fringe urchins at Sydney (Figure 5.2,3). At Sydney and Eden the urchins from the Fringe had a greater store of nutrients in their gonads at the beginning of the May gametogenic period and this resulted in a greater production of gametes compared with their conspecifics at the Barrens. This enhanced condition at the onset of breeding supported prolonged gamete storage and extended the spawning period of the Fringe urchins at Sydney in both years and at Eden in 1994 (Figure 5.2,3). Ulladulla differed from the other locations in having a similar breeding period at the Barrens and Fringe in both years (Figure 5.3).

Morphometry of gonad growth

The relationship between test diameter and reproductive output by *Centrostephanus rodgersii* assessed when the gonads were at their heaviest shows that gonad weight increased with increasing diameter (Figure 5.4a, b). There was considerable variation in gonad weight with the r^2 values for the Fringe and Barrens data sets being 0.39 and 0.63, respectively.

There was no significant relationship between size and relative yield in urchins from either the Barrens ($r^2 = 0.005$, ns.) or Fringe habitats ($r^2 = 0.004$, ns.). At maximum gonad output the GRR of the urchins from the Barrens habitat were on average less than 10%, while those from the Fringe were greater than 10% (Figure 5.4a, b, Table 5.1).

Differences among locations in gametogenesis and spawning

At all four locations the increase in GRR in May was due to an accumulation of nutritive tissue followed by mobilisation of stored nutrients for gametogenesis. The marked inverse relationship between nutritive

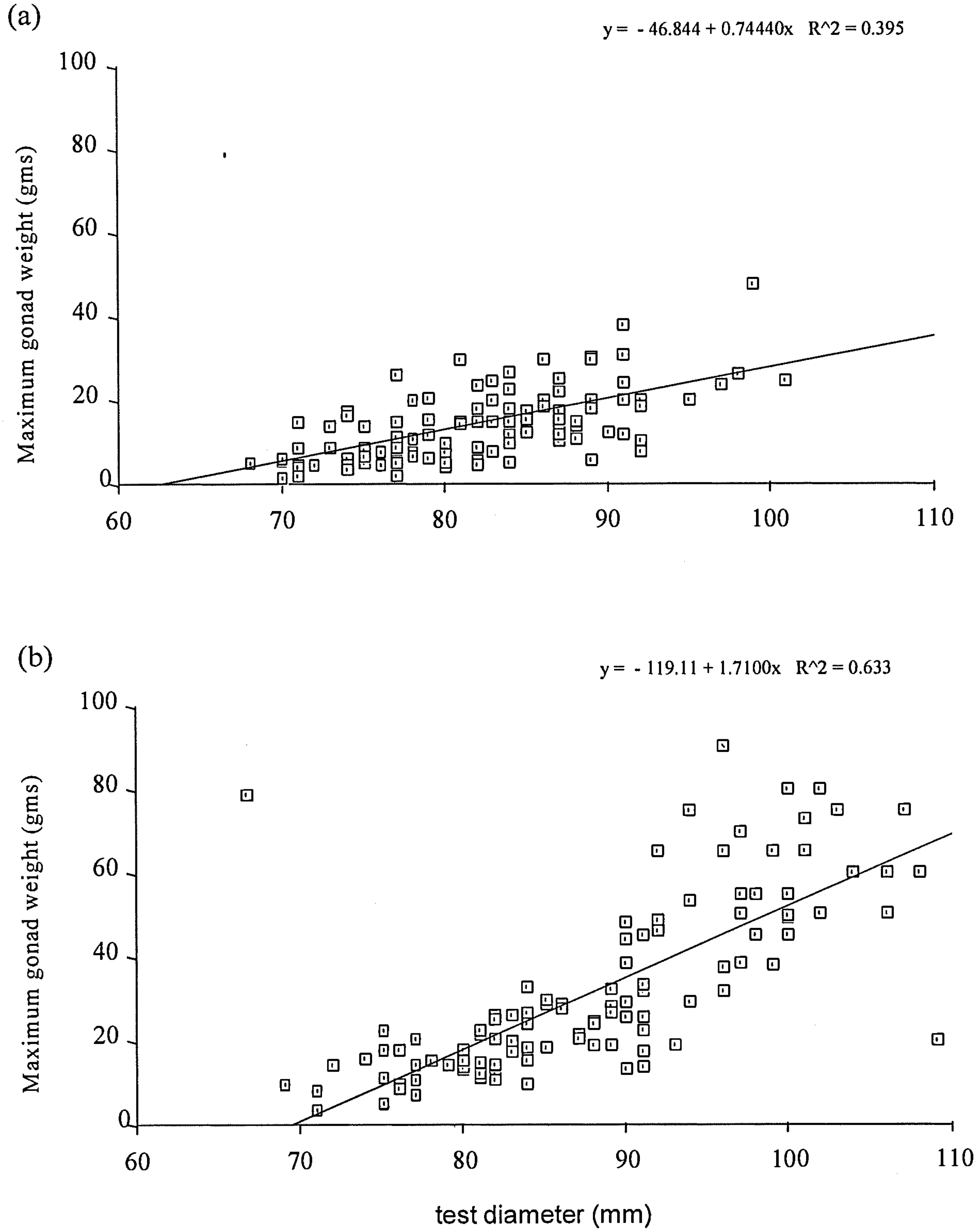


Figure 5.4 Relationship between maximum gonad weight and test diameter of *Centrostephanus* across all locations at the Barrens (a) and Fringe (b) habitat. a. r2 = 0.39; b. r2 = 0.63.

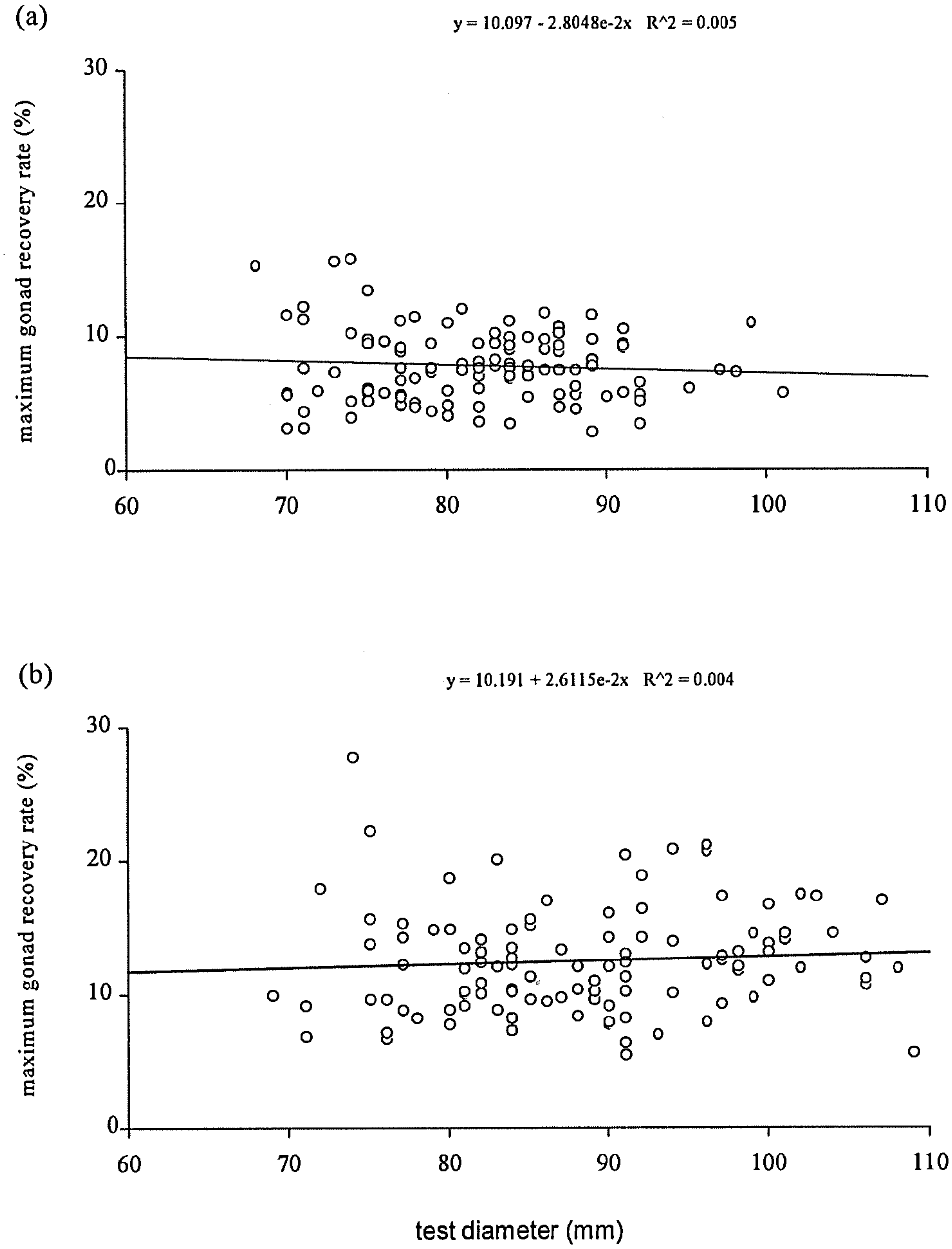


Figure 5.5 Relationship between maximum gonad recovery rate and test diameter of *Centrostephanus* across all locations at the Barrens (a) and Fringe (b) habitat. a. r2 = 0.005; b. r2 = 0.004.

Location	Latitude (°S)	Sea temperature range (°C)			Breeding period
		Annual	June/July		
			1994	1995	
Solitary Islands	30	19.4-27.3	20.4-21.2	20.5-20.8	June-July
Sydney	33	15.9-25.8	17.8-18.7	16.4-18.7	June-Sep./Oct.
Ulladulla	35	14.3-25.6	16.2-17.8	14.6-18.2	June-Sep.
Eden	37	12.3-23.7	13.5-17.0	13.3-17.5	June-Oct./Nov.

Table 5.1 *Centrostephanus rogersii.* Sea temperature range (annual and during spawning period) and breeding patterns at four locations along the New South Wales coast.

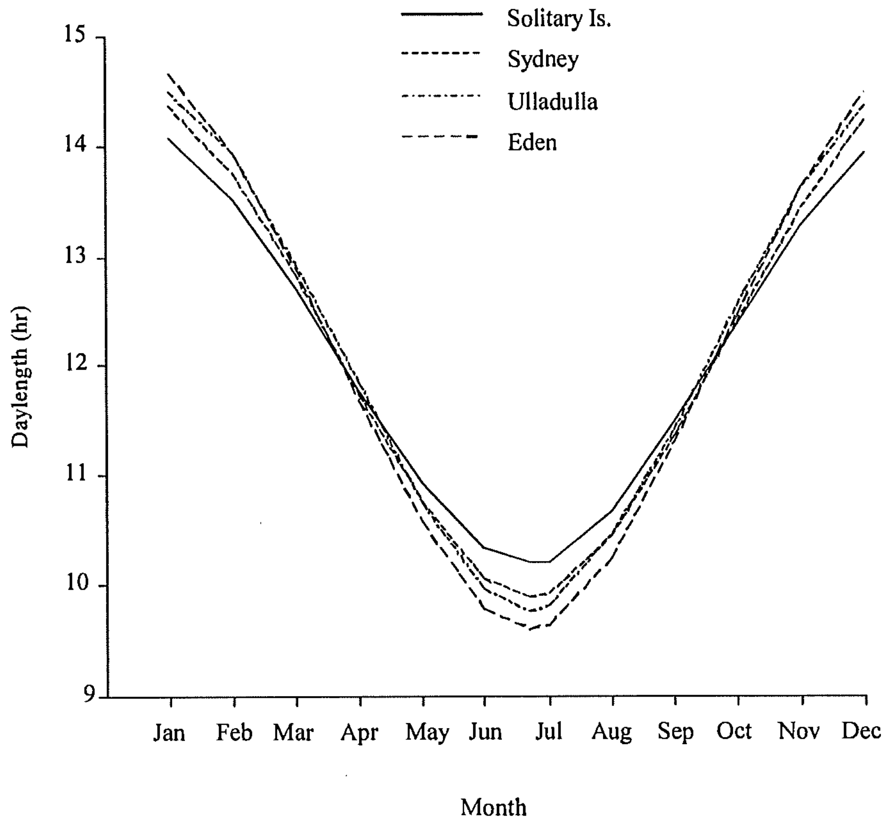


Figure 5.6 Annual day-length cycles at the Solitary Islands, Sydney, Ulladulla and Eden.

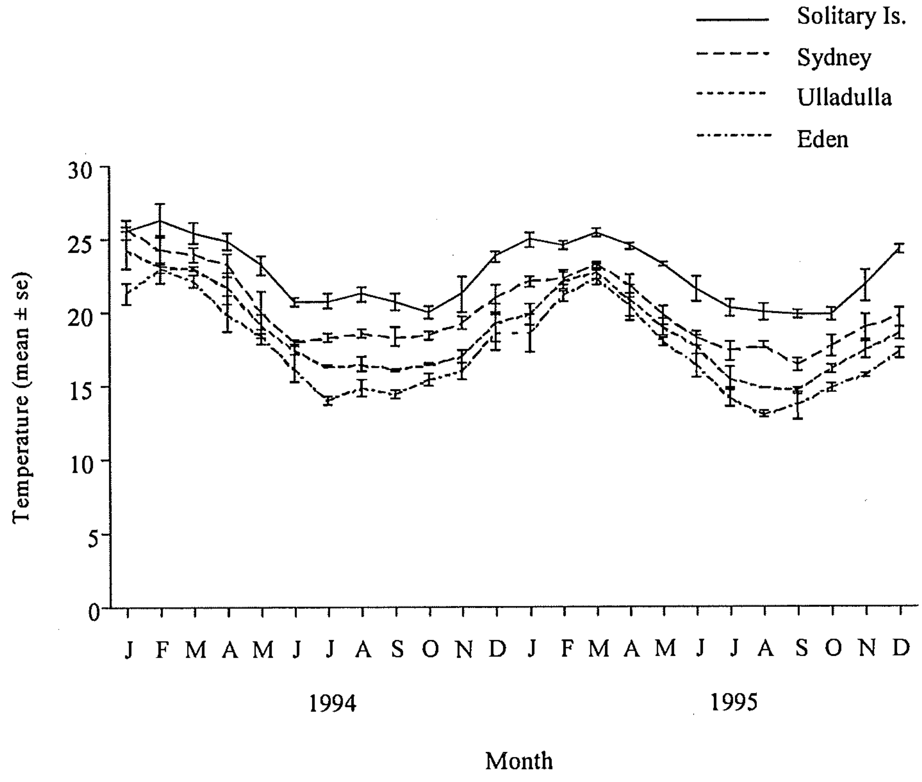


Figure 5.7 Mean sea temperatures at the Solitary Islands, Sydney, Ulladulla and Eden, January 1994 to December 1995.

tissue and gamete development was particularly clear at the Solitary Islands where the gonads exhibited sharply synchronous development of a single cohort of gametes. In contrast, stored nutrients were utilised over a longer time frame at the southern locations in association with ongoing gametogenesis. At the Solitary Islands gamete development was completed by mid June in 1995 followed by complete spawn-out. In contrast, at Sydney, Ulladulla and Eden mature gametes were provided by the germinal epithelium in late June and early July.

The onset of spawning occurred over a remarkably similar time frame at the four locations (Figure 5.1-3). At all sites, *Centrostephanus* had mature gametes by early June followed by the onset of spawning in mid to late June (Figure 5.2-3). At the northern end of its range spawning lasted only one month while at the southern end of its range there was an extended five to six month breeding period (Figure 5.3). The presence of mature gametes from August onwards at the southern sites was due to storage.

There was only a 35 minute difference in day length between the most northern and southern sites on the shortest day of the year (Figure 5.6). On 21 June, day lengths at the Solitary Islands, Sydney, Ulladulla and Eden, were 612, 594, 586 and 577 minutes, respectively. The increased tempo of gametogenesis in May occurred after the inflection point in April when day length became shorter than night length. At all locations *Centrostephanus* came into breeding condition near the winter solstice and spawning coincided with increasing day length.

Changes in sea temperature in 1994 and 1995 were smallest at the Solitary Islands and greatest at Eden, the most southern location (Figure 5.7). At the Solitaries, Sydney, Ulladulla and Eden, annual sea temperatures ranged from 19.4 - 27.3°C, 15.9 - 25.8°C, 14.3 - 25.6°C, 12.3 - 23.7°C, respectively (Table 5.2). The biggest difference was in minimum sea temperature with a 7°C difference between the Solitaries and Eden (Table 5.2). Maximum temperature at these locations differed by 3.6°C. Rapid oocyte growth in May coincided with decreasing temperatures and spawning in June and July coincided with the coldest temperature for the year (Table 5.2). In 1995 spawning at the Solitary Islands was bracketed by temperatures of 20.5 °C and 20.8 °C. The major June-July spawning period at Sydney coincided with temperatures ranging from 16.4 °C to 18.7 °C. At Ulladulla and Eden this spawning period coincided with temperatures ranging from 14.6 °C to 18.2 °C and, 13.3 °C to 17.5 °C, respectively. Across all locations, initiation of spawning coincided with temperatures between 13.3 °C and 20.8 °C (Table 5.2).

Gonad maturity stage	Months	GRR (%)			Commercial quality
		Sydney	Ulladulla	Eden	
Partly spawned/spent	Oct-Dec	4.1-9.2	4.1-5.7	6.2-7.8	low
Spent/recovery, growing	Jan-May	7.0-11.5	6.1-8.2	6.1-8.6	medium-high
Mature/partly spawned	June-Sep.	5.0-8.2	1.5-9.4	5.2-13.4	medium-low

Table 5.2 Centrostephanus rogersii. Seasonal trends in gonad maturity (stages detailed in King et al. 1994), GRR and expected commercial value of roe in urchins from fringe habitats in Sydney, Ulladulla and Eden.

5.5 Discussion

Gametogenesis in *Centrostephanus* was typical of diadematoids, with a marked temporal separation of gonad growth due to enlargement of the nutritive phagocytes followed by mobilisation of the reserves during an intense period of relatively synchronous gametogenesis prior to spawning (King et al. 1994, O'Connor et al. 1978, Pearse and Cameron 1991). The onset of vitellogenesis in May was well-illustrated by the appearance of a rapidly growing cohort of oocytes in the ovaries of urchins from the Solitary Islands. Vitellogenesis in *Centrostephanus* takes approximately one month, similar to that reported for other echinoids (Gonor 1973, Byrne 1990). At the Solitary Islands the pre-spawning gametogenic period involved the entire cohort of gametes and marked the end of gametogenesis for the season. Gametogenesis was less synchronous at Sydney, Ulladulla and Eden where gamete development continued through late June and early July.

The major difference in reproduction by *Centrostephanus* along the coast of New South Wales was in the duration of spawning. A rapid increase in gonad weight followed by complete spawn-out over a few weeks is characteristic of reproduction by *Centrostephanus* at the Solitary Islands (O'Connor et al. 1978). Spawning at this most northern site in 1995 was finished by July. In contrast, spawning by *Centrostephanus* at Sydney, Ulladulla and Eden in June and July 1994-5 was partial. At these sites gamete storage particularly by the Fringe urchins ensured that spawning continued for several months. The longest breeding period was recorded at the most southern location at Eden.

An increase in the tempo of gametogenesis in May and onset of breeding in June-July occurred in parallel at all locations. This reproductive synchrony in *Centrostephanus* is consistent with gametogenesis and spawning being cued by exogenous factors operating across all the populations studied. At all locations, spread over nine degrees of latitude, the most consistent and most likely factor to entrain these reproductive events is photoperiod. Initiation of gametogenesis appears to be triggered by decreasing day length and probably occurs in April when the days becomes shorter than nights (as shown for the urchin *Strongylocentrotus purpuratus*, Pearse 1981). The gonads reached their maximum development just prior to the winter solstice and spawning occurred shortly thereafter. For *Centrostephanus* short days and lunar conditions coinciding with the solstice appear likely proximate factors which might cue the onset of spawning across its range. A similar reproductive pattern occurs in widely separated populations of *S. purpuratus* along the west coast of North America (Pearse 1981). Detailed field and laboratory studies demonstrate that gametogenesis in *S. purpuratus* is controlled by seasonally changing photoperiod with little or no influence by temperature (Gonor 1973, Pearse et al. 1986, Bay-Schmith and Pearse 1987).

Although gametogenesis and spawning coincided with decreasing sea temperature, this factor would not provide a uniform cue to promote reproductive synchrony at all locations. In light of the contrasting winter temperatures experienced by northern and southern populations, it is unlikely that *Centrostephanus* has a critical temperature for initiation of breeding, as suggested for other sea urchins (Fenaux 1968, Byrne 1990). Temperature however, may influence gamete storage and duration of spawning. The differences observed among locations in the present study suggest a latitudinal trend in breeding by *Centrostephanus* with a one month spawning period at its northern limit and a five to six month spawning period at its southern limit. Cool temperature has been suggested to play a role in extending gamete storage and spawning by deep water populations of *S. purpuratus* (Leahy et al. 1981).

Many diadematoids reproduce with a monthly pattern of gametogenesis and spawning which are under lunar control (Pearse 1970, Kennedy and Pearse 1975, Lessios 1984, Pearse and Cameron 1991). For *Centrostephanus*, synchronous growth of the cohort of gametes in urchins from the Solitary Islands reflects the diadematoid pattern. The possibility that gametogenesis at this location is influenced by the lunar cycle would have to be assessed by weekly examination of the gonads with respect to the lunar phases in May and June. If reproduction at Sydney, Ulladulla and Eden is influenced by the lunar cycle, the prolonged pattern of gametogenesis at these sites would make this difficult to discern. Within the annual breeding period of its congener, *C. coronatus*, gametogenesis and spawning occur with a monthly pattern (Kennedy and Pearse 1975). Extended spawning by *Centrostephanus*, however, was due to prolonged gamete storage and not due to a monthly renewal of gametogenesis.

Growth of sea urchin gonads is highly sensitive to food quantity and quality (Ebert 1968, Vadas 1977, Pearse 1981, Andrew 1986, Byrne 1990, Pearse and Cameron 1991). As seen here for *Centrostephanus*, inhabiting food-poor Barrens areas comes at a cost to reproduction. At the Solitaries, Sydney and Eden, the Barrens urchins had a lower reproductive output compared with those in the Fringe. The underlying cause of the lower GRR and shorter breeding period *Centrostephanus* at the Barrens was most probably the low abundance of macroalgal food which resulted in the urchins having a smaller store of nutrients at the onset of gametogenesis. The property of the gonads as the major nutrient storage organ in sea urchins and the ability of the nutritive phagocytes to respond to food of differing quality is utilised by fisheries and aquaculturalists to enhance gonad output through control of diet (de Jong-Westman et al. 1995, Goebel and Barker 1997, Lawrence et al. 1997). This has prompted research to identify optimal natural diets and formulation of artificial diets for sea urchins with the aim to achieve maximum yield of high quality gonads that have the yellow-orange colour preferred by the market (de Jong-Westman et al. 1995, Goebel and Barker 1997, Lawrence et al. 1997). For aquaculture, the potential for re-programming reproduction in urchins through alteration of photoperiod creates the potential to produce out-of season crops of gonads (Pearse et al. 1986, Bay-Schmith and Pearse 1987). For *Centrostephanus* it appears that manipulation of summer photoperiod may enhance gonad growth out of season (see also *Strongylocentrotus purpuratus*, Pearse et al. 1986).

The suitability of *Centrostephanus rodgersii* for commercial harvest in New South Wales depends on the reproductive cycle (Table 5.3). The roe are optimal for harvest when they have a firm texture during the recovery and growing stages. *Centrostephanus* with recovering and growing stage gonads were abundant at Sydney, Ulladulla and Eden from January to May. Within this period, yields would be greatest in late March to early May prior to the mobilisation of nutrient stores for gamete production. During May the nutritive cells shrink and the gonads soften as the number of gametes increases. Although the short, synchronous reproduction of *Centrostephanus* at the Solitary Islands means that they have marketable gonads for two to three months longer, it is unlikely that a commercial fishery could be sustained because of comparatively low densities of urchins and the high conservation status of these reefs.

With respect to the importance of obtaining a high yield of gonad per urchin, it is clear that harvesting of *Centrostephanus* in Fringe habitats would maximise the return to the fishery. Although there was considerable variation among urchins, the GRR in the Fringe habitat was at or above 10%. In contrast the recovery rates obtained for urchins from the Barrens habitat were generally less than 10%, indicating these habitats may not yield sufficient roe to warrant harvesting.

5.6 Conclusions

The greatest yields of high quality roe from *Centrostephanus* were obtained from the Fringe habitat in summer and autumn. Although post-harvest treatment of the roe may enhance roe quality to some extent, it is probable that roe harvested from the Barrens habitat at any time and from algal-dominated habitats at other times will be either of poor quality and/or have an uneconomically low recovery rate. We recommend fishing practices that promote high yield of quality roe, such as paying divers for the weight or quality of roe harvested rather than the weight of sea urchins landed. Paying by weight of whole sea urchin will promote larger harvests of poorer quality roe.

Developing A Management Plan for the Urchin fishery

The development of the sea urchin roe fishery should be viewed within the larger context of the abalone fishery. The total harvest of sea urchins from New South Wales has been small (see Figure 1.2). NSW Fisheries is therefore in the enviable position of being able to build a fishery from an unfished condition. The development of a management strategy for a sea urchin roe fishery in New South Wales must accommodate the strong ecological interactions between urchins, abalone and other species on shallow rocky reefs.

A first step in developing a management strategy for the sea urchin fishery will be the formulation of a management plan with goals and objectives that reference not only the roe fishery, but also the abalone fishery. If the management plans are not complimentary then the roe fishery may develop goals inconsistent with those existing for the abalone fishery. However, given the results presented in this study, it should be possible to develop plans to manage these fisheries in a way that enhances both.

There is currently insufficient information to sustain an assessment model for a *Centrostephanus* roe fishery; there are no times series of catch and effort and there are currently no estimates of key population parameters such as growth, mortality and size/age-specific fecundity. Further, the fertilization and recruitment dynamics shown to be important in other sea urchin fisheries (Pennington 1985, Ebert and Russell 1989, Botsford et al. 1993, Levitan and Sewell 1998) remain poorly described for *Centrostephanus*. These facts, overlaid with the strong ecological role this species plays and its relatively sedentary habit, suggest that an experimental approach should be taken, at least in the short term. The foundation for this approach has been laid with the limitation, in 1997, of fishing to 5 of the 83 subzones used for reporting catch and effort in the abalone fishery. Harvesting only in these subzones will maximise the ecological impact of the fishery in small areas of the coast. Fishing in these subzones may be rotated (e.g. Bradbury 1991, Botsford et al. 1993, Caddy 1993, Pfister and Bradbury 1996, Pfeifer 1997) and different harvesting regimes tested. Some subzones should remain closed to fishing (e.g. Caddy 1989, Sims 1992, Quinn et al. 1993, Rogers-Bennett et al. 1995, Holland and Brazee 1996).

It is important to remember, however, that this experimental approach to management must be pursued in a careful manner. To quote McConnaha and Paquet (1996), any adaptive management approach should be:

> "focused through the use of an explicit learning plan that links goals, uncertainties, and actions. Such a plan presents an organised approach to learning and could prevent the diffusion of activities that can occur under the banner of adaptive management"

Future research should concentrate on gathering the necessary biological information, along with fishery independent surveys of the resource to support assessment modelling. These data, along with catch and effort data gathered on a small spatial scale will allow the management of the fishery to develop in sophistication as the information base grows. Points of focus in this research include:

- Estimates of age and growth. Preliminary analysis of tagged urchins recovered after one year at liberty suggest that *Centrostephanus* may reliably be aged using lines on the teeth (N.L. Andrew and C. Blount unpublished data). Variability in growth among subzones and at larger scales is likely to have a large influence on the period of rotation in the fishery (e.g. Caddy and Seijo 1998).

- Estimates of size-specific fecundity, mortality and growth for use in per-recruit modelling. The possibility of imposing a maximum size limit on the fishery may reduce the impact of Allee effects (Pennington 1995, Levitan and Sewell 1998).

- Gathering small-scale catch and effort statistics, possibly even within the sub-zones. Catch-effort statistics gathered at small and appropriate scales are an important source of information in the assessment of fisheries, including those for species with highly aggregated patterns of distribution (Keesing and Baker 1998, Worthington et al. 1998). Developing technology may allow the collection and analysis of catch and effort data on very small spatial scales (e.g. Boyd et al. 1996, Rodgers and Rouse 1997). The possibility of implementing small scale management to this resource should be carefully considered, particularly in the developing phase of the fishery.

- As the time series of data develop, management strategies (e.g. constant catch or constant escapement) and consequent assessment models can be formulated (e.g. Quinn et al. 1993, Pfister and Bradbury 1996, Lai and Bradbury 1998).

This course of research, along with information on the processes determining roe quality (e.g. differences among habitats, densities and diets) will maximise the chances of long-term sustainability in this fishery.

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6. Recommendations and Implications

6.1 Benefits

The benefits from this research have flowed to the NSW Abalone fishery in several ways. First, the fishery now has annual stock assessments. These assessments are presented at Industry Open Days, to ABMAC and, finally to the NSW TAC committee. The TAC committee has used the annual assessment reports as their primary source of information in setting annual TACCs. The management plan for the NSW abalone fishery is based on objectives and performance criteria developed as part of the present research program. These performance criteria determine priorities for the on-going assessment program. The development of stock assessment methods for the NSW abalone fishery will benefit other abalone-producing states. The modelling framework and assessment methods have been presented to researchers from all state agencies.

The demonstration of the impacts of sea urchins on the abalone fishery has led to the establishment of management structures to develop the sea urchin fishery in such a way as to enhance the abalone fishery. More indirect benefits will flow from a greater understanding of the ecological systems within which these fisheries operate and through the study of interactive fisheries. The results of this research have been presented in international scientific forums and are being published in the primary scientific literature.

The recent extension of *Centrostephanus* into north eastern Tasmanian waters means that the greater ecological understanding of the relationship between this species and blacklip abalone will be of benefit to Tasmania.

6.2 Intellectual Property

No patents emerged from this research. All results will be published by the NSW Fisheries Research Institute in reports, public domain scientific journals and presented at Industry seminars and scientific conferences.

6.3 Further Development

The present FRDC-funded project was used as leverage to develop a related stock assessment program for the NSW abalone fishery. This latter program was partially funded by the commercial divers, beginning in 1993, and has provided annual stock assessments since (Andrew et al. 1996a, 1996b, Worthington et al. 1997, 1998). The stock assessment program is now fully funded by the NSW commercial abalone industry. Related research on the sea urchin fishery is funded by NSW Fisheries and the NSW commercial abalone Industry and research on abalone enhancement is funded by FRDC.

As spin-off from the present study, a PhD student (David Ward) at the University of Wollongong is currently studying the fisheries biology of turban snails (*Turbo torquata*). Turban snails are part of the

NSW abalone fishery but are under-utilised, partially because of conservative MLS set many years ago. The focus of this research is an investigation of the appropriateness of the current size limit for this species.

6.4 Staff

The following NSW Fisheries staff were directly employed on this project:

Neil Andrew	Principal Investigator
Duncan Worthington	Senior Technical Officer
Penny Brett	Technical Officer
Nokome Bentley	Technical Officer

Other NSW Fisheries staff, contributing to the project but not directly funded were:

Ron Avery	Technical Officer
Craig Blount	Technical officer
Rowan Chick	Technical Officer
Peter Gibson	Technical Officer
Elizabeth Hayes	Technical Officer

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8. Appendix 1.

Examples of Maps generated for 60 sites in southern New South Wales. See following page for key to habitat representation.

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Key to colours used for each habitat type

🟥	Barrens
🟪	*Eclonia* Forest
🟪	Turf
🟩	*Phyllospora* Forest
🟧	*Pyura*
🟥	*Durvillea*
🟩	Fringe
🟨	Sand
🟫	Intertidal
⬜	Gravel
🟦	White Water

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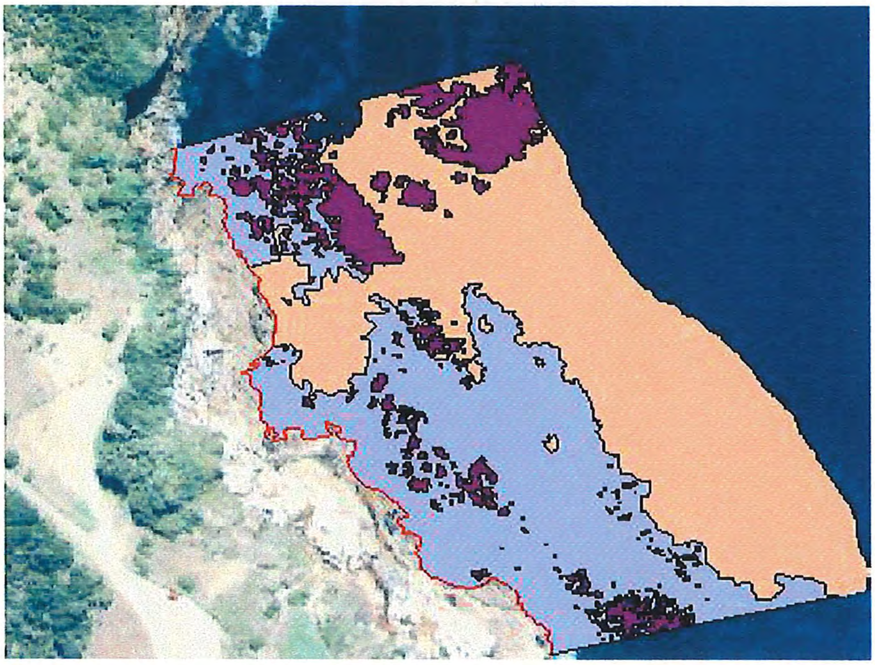
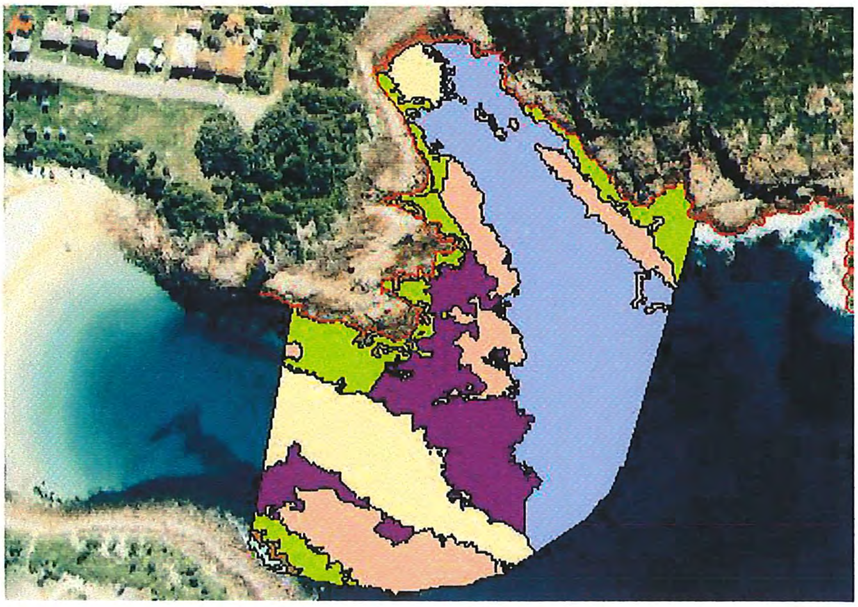
Cape Solander, Sydney



Boat Harbour, Port Stephens



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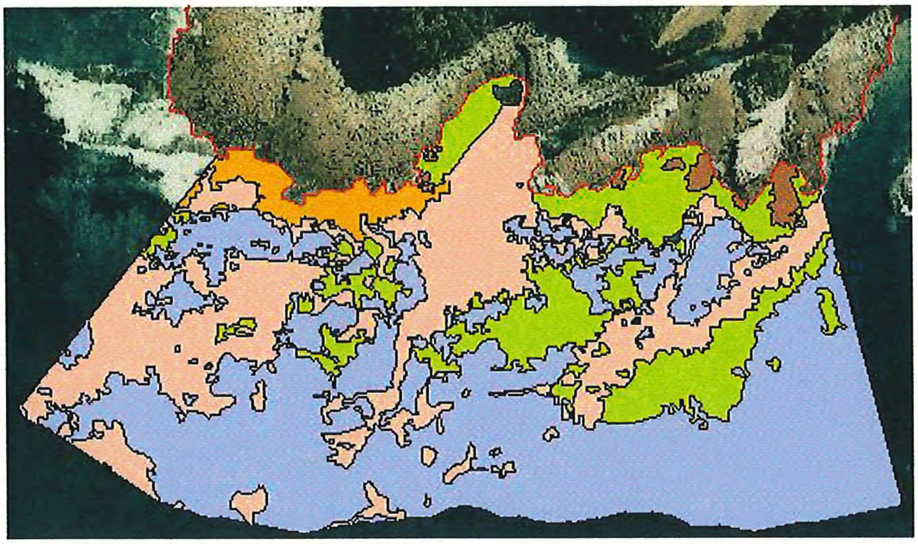
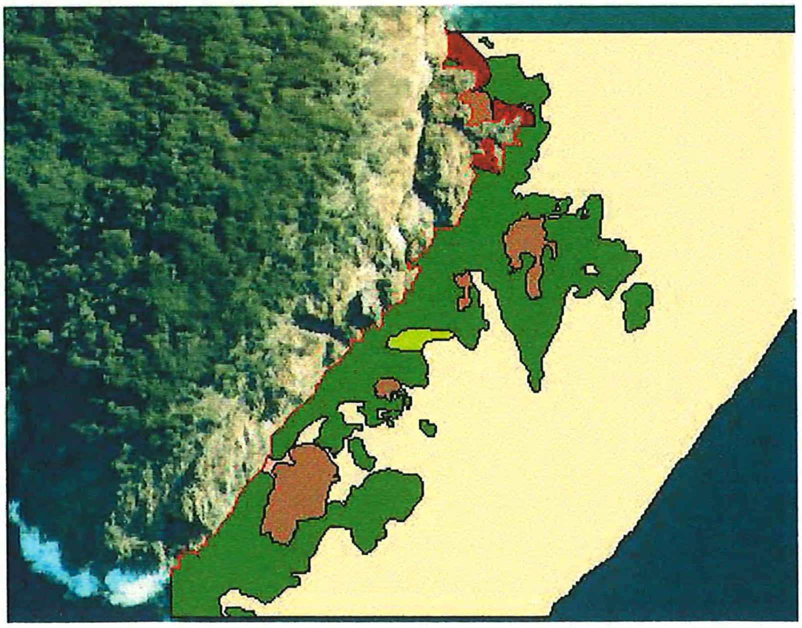
Mary Cove, Jervis Bay



Turingal, north of Merimbula



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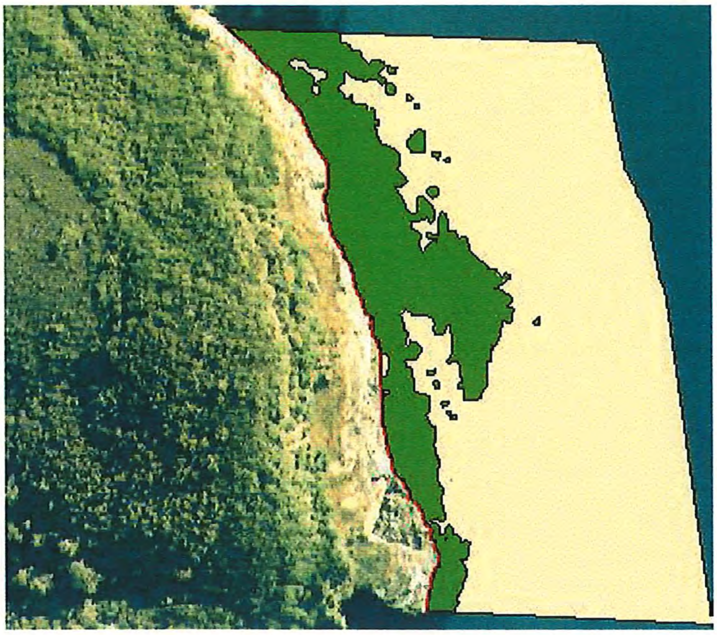
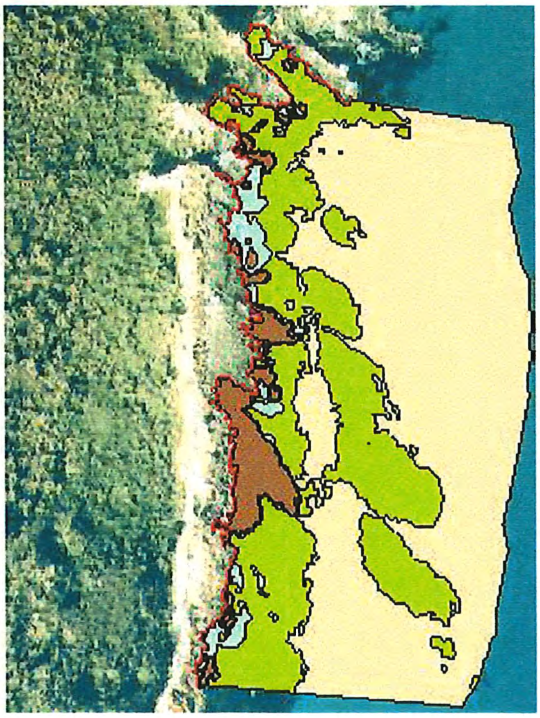
Nadgee site 2, south of Eden

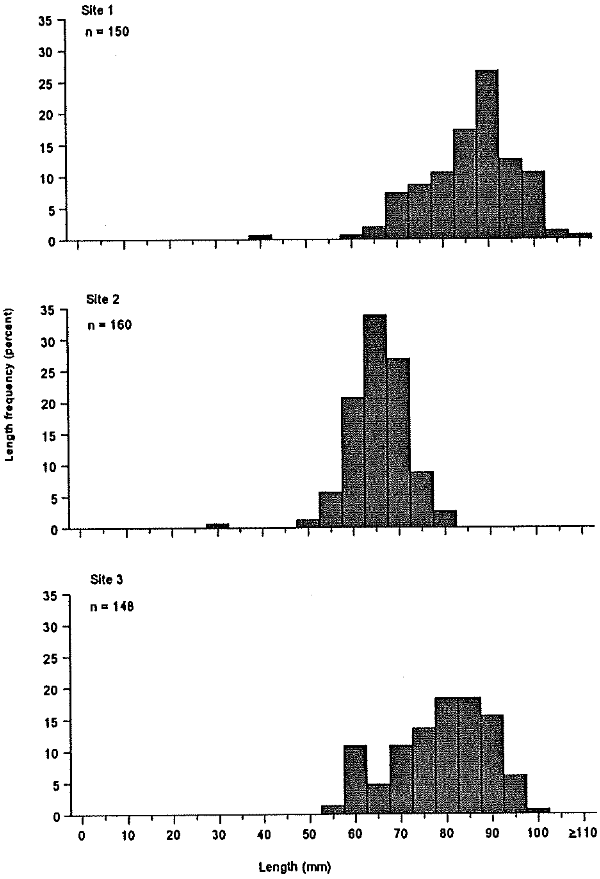


Nadgee site 2, south of Eden

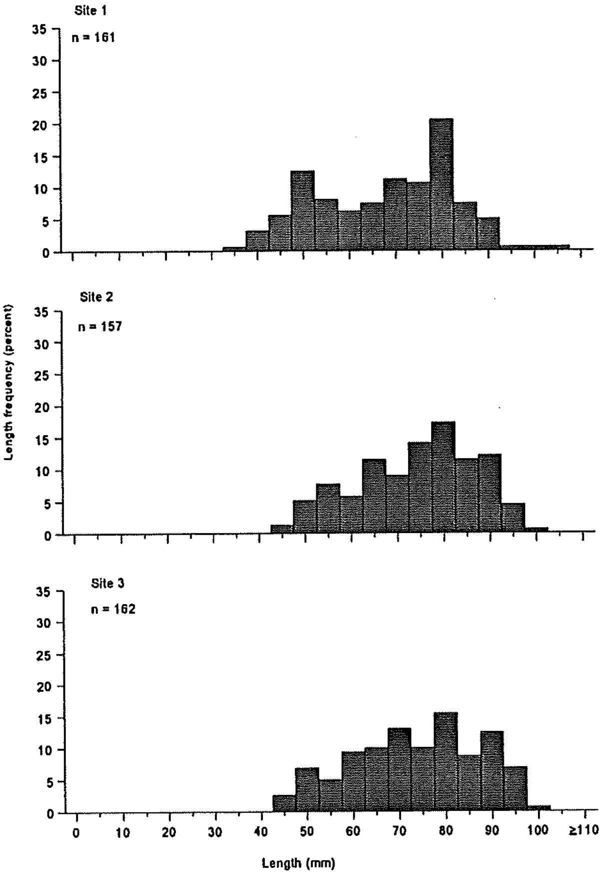


9. Appendix 2.

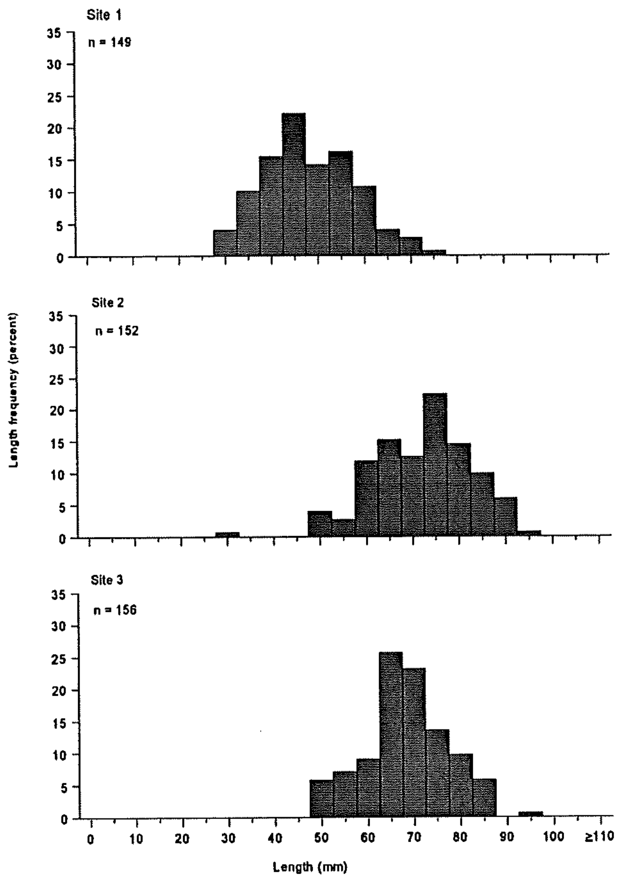
Size-frequency distributions of Centrostephanus from 27 sites clustered in 9 locations in southern New South Wales. Sample sizes as indicated.



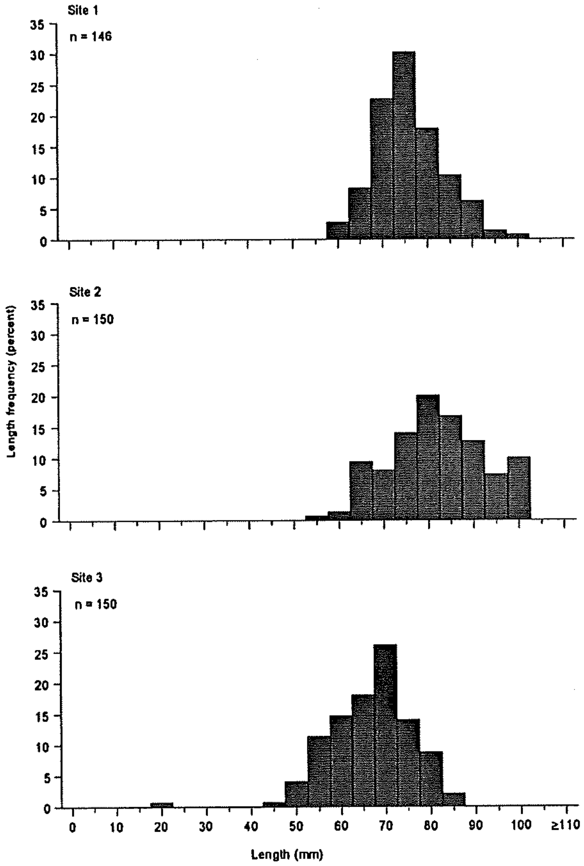
Appendix 2a) Length frequency distributions of *Centrostephanus* for three sites in Port Stephens.



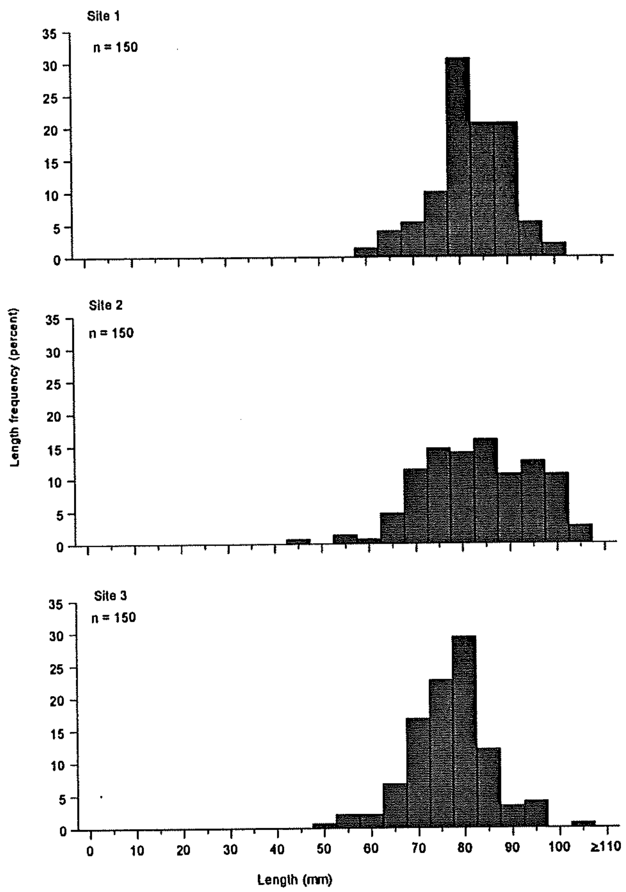
Appendix 2b) Length frequency distributions of *Centrostephanus* for three sites in Sydney.



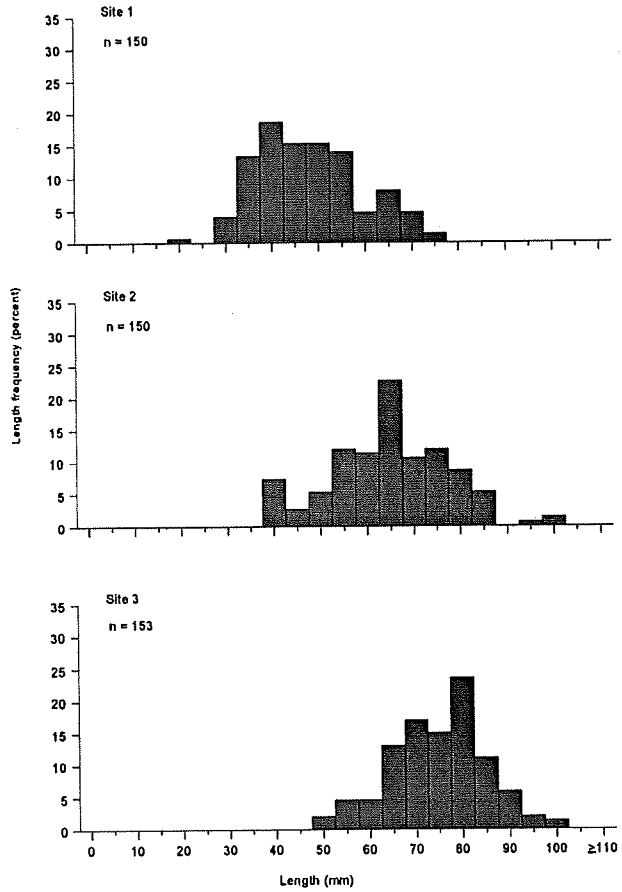
Appendix 2c) Length frequency distributions of *Centrostephanus* for three sites in



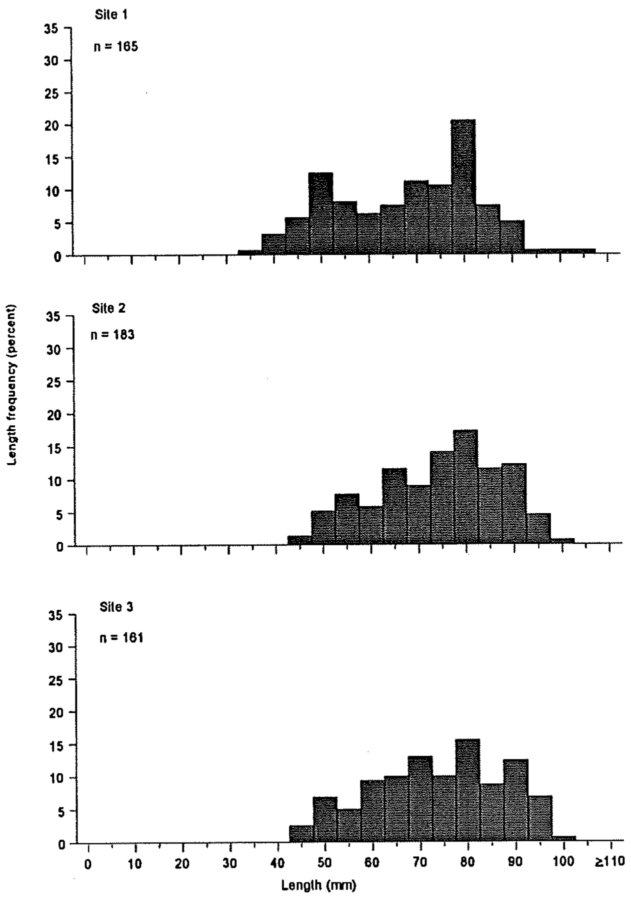
Appendix 2d) Length frequency distributions of *Centrostephanus* for three sites in Ulladulla.



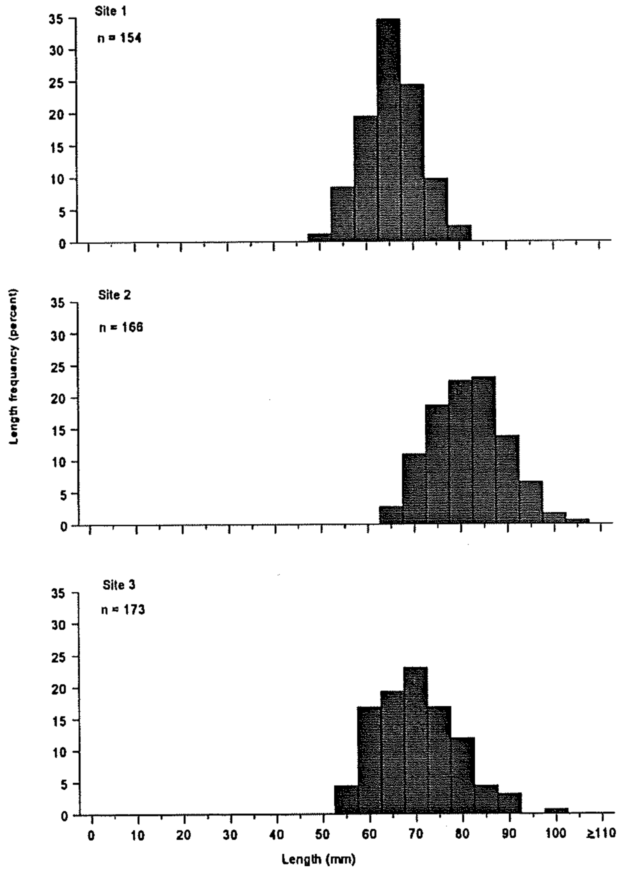
Appendix 2e) Length frequency distributions of *Centrostephanus* for three sites in Jervis Bay.



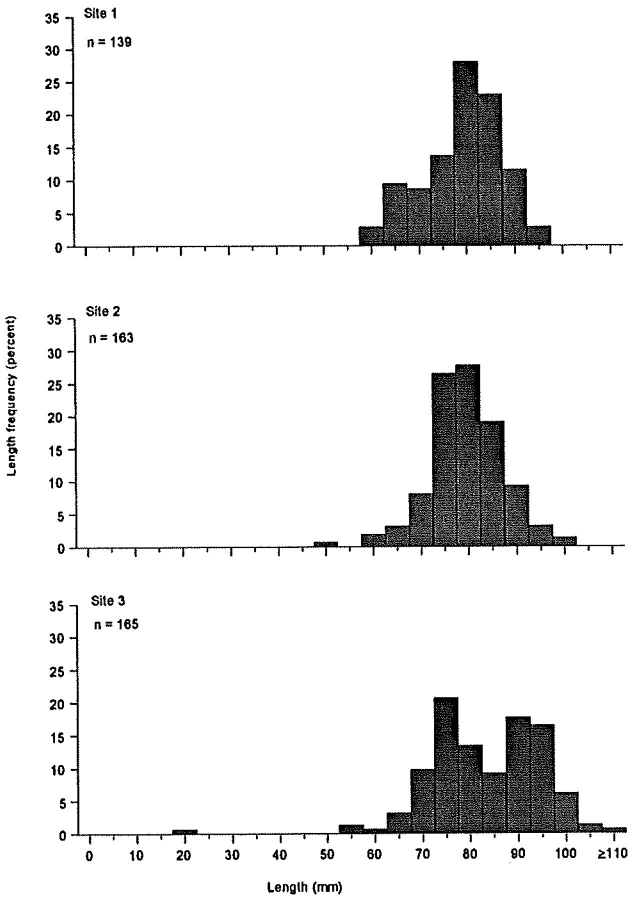
Appendix 2f) Length frequency distributions of *Centrostephanus* for three sites in Batemans Bay.



Appendix 2g) Length frequency distributions of *Centrostephanus* for three sites at Haycock Point.



Appendix 2h) Length frequency distributions of *Centrostephanus* for three sites in Eden.



Appendix 2i) Length frequency distributions of *Centrostephanus* for three sites in Disaster Bay.

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