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MONITORING OF TASMANIAN INSHORE REEF ECOSYSTEMS

An assessment of the potential for volunteer monitoring programs and summary of changes within the Maria Island Marine Reserve from 1992-2001

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AUSTRALIA

Natural Heritage Trust

Helping Communities Helping Australia

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Summary

A grant from the Monitoring Program of Coasts and Clean Seas to the Tasmanian Aquaculture and Fisheries Institute enabled the continuation of a long-term monitoring program examining changes within Tasmanian inshore reef ecosystems associated with marine protected areas. This work extended the program over the period autumn 1999 to autumn 2001. An additional component of this grant included an assessment of the potential use of community based volunteers to monitor key aspects of inshore reef ecosystems.

Analysis of the results of the long-term monitoring was restricted to notable changes occurring within the largest Tasmanian coastal MPA at Maria Island. After nine years of protection at the Maria Island MPA, mean size of lobsters was significantly greater than in the surrounding fished waters and still increasing. The total abundance of lobsters now appeared to be relatively stable following a threefold increase during the first five years of protection. The mean size of abalone continued to slowly increase, while the abundance of abalone appeared to have declined by 50% over the nine years.. This decline may be due to several causes, including an increase in predator density (eg. lobsters and large fishes) and increased competition resulting in delayed emergence or higher mortality. Urchin abundance has declined by up to 40% and may also be related to increased levels of predation. The monitoring has revealed shifts in species distributions that may be related to long-term oceanic cycles, and patterns in the invasability of an introduced macroalgae *Undaria pinnatifida*.

Monitoring by community based volunteers demonstrated that with a limited level of training on species identification and quantitative techniques, volunteer groups could provide data that were sufficiently reliable to characterise a “place” at a particular point in time with respect to the assemblage present. While the results of individual divers were highly variable, pooling results of volunteers at each location produced a result similar to that obtained by skilled observers.

At an individual species level, or for estimation of the mean size of individual species, the volunteer data was less reliable than that provided by skilled researchers, however, with sufficient training volunteer data may show significant improvement. An essential component of community involvement is to ensure adequate training and feedback for all divers, that events are well coordinated, and adequate post dive support is on hand to ensure species are identified and recorded accurately and legibly.

With good training and professional facilitation of volunteers, community based monitoring may provide data with sufficient reliability to detect substantial shifts in reef ecosystems. However, unless this is organised on a large scale by volunteer organisations, logistical constraints mean data obtained by this method are likely to be substantially less cost effective or reliable than that obtained by research organisations with adequately trained staff.

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1. Introduction

Understanding patterns in the distribution and relative abundance of reef flora and fauna, and the basic mechanisms of reef ecosystems is an essential precursor for coastal management and planning. However, the baseline data and information on temporal trends (natural or otherwise) necessary for informed decision making is generally scarce in temperate Australia, limiting our ability to effectively manage and conserve marine systems. Currently conservation planning (particularly for MPA's), and the management of many reef fisheries and habitats, is based on insufficient data to be certain the desired outcomes can be achieved. This lack of information has been identified as a priority area for future research by the Tasmanian State of Environment Report (SDAC 1996), the Commonwealth State of the Marine Environment Report (Zann and Kailola 1995) and the Tasmanian NHT Unit. Priority areas identified by the NHT unit include monitoring of coastal habitats and overcoming our inadequate understanding and monitoring of natural systems. Many state policies and plans, including the State Coastal Policy, Tasmanian Fisheries Management Plans and the Tasmanian MPA Strategy (Marine and Marine Industries Council 2001) require that information on the current status of coastal marine biota is effective, however this information is currently lacking.

The required information would provide a greater understanding of a range of issues including:

- the sustainability of individual fisheries, the ecological impacts of current fishing practices,
- the extent that natural variability alters coastal marine communities,
- the impact of introduced marine species,
- the extent of habitat degradation,
- the effectiveness of MPAs and MPA design, and
- the most suitable MPA locations for achieving desired conservation outcomes.

Several recent studies in Tasmanian waters have provided at least a starting point for this process. Barrett *et al.* (2001) and Jordan *et al.* (2001) have respectively mapped habitats within the Bruny Bioregion and the Derwent Estuary, providing a baseline for change to be detected at the habitat level of classification within these areas, and providing detailed information at the habitat level for MPA planning.

In a long-term study examining the effects of the creation of marine reserves in Tasmanian waters, Edgar and Barrett (1997,1999) have documented changes in the biota of the reserves in the first five years (1992-1997) since their protection from fishing. This study identified a significant recovery of the reserves following the cessation of fishing and identified significant management issues, both within and outside the reserves, that need addressing. One example is that bastard trumpeter, a favourite fish of recreational gillnet fishermen, increased substantially in numbers within the Maria Island reserve but did not increase outside the reserve. A similar result was found with rock lobster where the biomass of adult lobsters was estimated to have increased by nearly ten times over this period. The results indicate that these species are heavily overfished on the Tasmanian east coast and management changes are needed to ensure long-term sustainability. Other findings included the lack of recovery of resident fish populations within the smallest reserves, indicating that the combination of small size and inappropriate boundaries was responsible for the loss of many fishes to fishing on the reserve boundary.

While the Tasmanian MPA study documented the value of surveying temporal patterns in reef ecosystems and MPAs, the five year period over which the study was conducted was insufficient to fully document the divergence between fished and unfished (MPA) locations or longer-term system-wide trends. As many of the common reef species are long-lived (eg. blue throated wrasse and purple wrasse - Barrett 1995, banded morwong -Murphy and Lyle 1999) changes may be expected to occur and accumulate over time periods well in excess of the five year period documented so far.

The first objective of the work reported here, funded by the Monitoring Program of Coasts and Clean Seas, was to allow the time series of the study to be extended by an additional three years (1999 to 2001). This substantially increases the length of the time (nine years) over which temporal trends can be examined. It also allows the divergence between protected and unprotected areas to be studied over a period of time sufficiently long to detect ecosystem changes due to the effects of fishing.

With the advent of the Natural Heritage Trust as the key funding source for marine conservation research and management in Australia there was a substantial shift towards community involvement at all levels. This is reflected in the development of programs such as Landcare, Coastcare, Fishcare and Waterwatch, programs that form partnerships between the community, industry and all levels of government to achieve shared goals. A component of the project reported here was to trial the development of a community-based volunteer monitoring program on temperate reefs (Reefwatch), to develop workable protocols, participation mechanisms and momentum. An additional component of this work was to examine the reliability of the data obtained by community volunteers by conducting comparisons with data collected by skilled researchers, and to refine methods such that community derived data was considered sufficiently reliable for use in coastal management, conservation planning and education. Comprehensive volunteer studies completed in countries like Belize (Mumby *et al.* 1995) and Tanzania (Darwall & Dulvy 1996) have enabled the development of MPA management plans and demonstrated that volunteer groups can play an important role in marine management.

Volunteer divers have been used with varying degrees of success from Tanzania (Darwell & Dulvy 1996) to the US (Schmitt & Sullivan 1996) and Australia (Musso and Inglis 1998), with the degree of success relating to the extent of diver training and the complexity of the task required. Volunteer temperate monitoring programs are also underway in Australia, with a temperate Reefwatch program now established in South Australia (www.reefwatch.asn.au). With the emergence in recent years of numerous and affordable detailed identification guides for temperate Australian reef fishes (eg Kuiter 1993, Gomon *et al.* 1994, Hutchins and Swainston 1986, Edgar *et al.* 1982), invertebrates (eg Edgar 1997, Shepherd and Thomas 1982, 1989, Shepherd and Davies 1997, Jansen 2000) and macroalgae (eg Christianson *et al.* 1981, Huisman 2000) the ability to train volunteers in underwater identification has substantially improved. Using these guides, volunteers could conceivably be involved in monitoring programs covering a wide range of biotic groups, particularly if the organisms censused were restricted to the more common and readily identifiable species.

The second objective of this study then was to trial a volunteer monitoring program on inshore Tasmanian reefs using a similar methodology to that currently used in the long-term Tasmanian MPA study and a methodology widely used in similar studies elsewhere (eg Lincoln-Smith 1989, Cole *et al.* 1990). The rationale for this was twofold. Firstly, well-trained community volunteers may be able to continue an invaluable time series of data in the future if funding was not available for the work to be conducted by research staff. Secondly, monitoring is required at a wide range of locations around the Tasmanian coastline to document patterns and changes on a regional basis. One urgent requirement is to document natural assemblages before introduced species such as the invasive macroalgae *Undaria pinnatifida* invade reefs on a widespread basis. Recent surveys within the Tinderbox marine reserve indicate that this species can form almost 100% canopy cover under favourable conditions. Monitoring at this scale is usually beyond the resources of research agencies but may be achievable if organised and conducted by enthusiastic volunteer groups such as the Tasmanian Marine Naturalists.

2. Methods

2.1 Annual surveys by trained staff

In the autumn of 1999, 2000 & 2001 staff from the Tasmanian Aquaculture and Fisheries Institute continued annual monitoring of Tasmanian MPAs and their external reference sites. In each autumn a total of 26 sites were surveyed by a core staff of three divers. The techniques used and locations surveyed are described in Edgar and Barrett (1997 & 1999) and are included here for completeness.

2.1.1 Sites examined

Biological changes that followed the declaration of marine reserves were quantified using underwater visual census at sites within reserves and at reference sites outside reserves that possessed a similar habitat type. The effects of reserves could only be distinguished from long term trends in Tasmanian coastal waters when changes in a reserve were found to be significantly larger or smaller than changes outside the reserve (Green 1979).

Our study concentrated on species associated with reefs because this habitat type is the most heavily targeted by inshore fisheries, and because many reef-associated species are site attached and so should recover relatively rapidly in marine reserves. By contrast, most open water and soft-bottom fishes are unlikely to remain in small marine reserves for sufficient time to receive adequate protection.

The four Tasmanian marine reserves investigated varied substantially in size, with ≈7 km of coastline protected from fishing in the Maria Island Marine Reserve, 2 km protected in the Tinderbox Marine Reserve, and ≈1 km in the Governor Island and Ninepin Point Marine Reserves. In order to maintain consistency of spatial scale, sites selected for monitoring (each ≈250 m across) were separated by a similar distance in all reserves (≈1 km between neighbouring sites). Approximately half of the total sampling effort was therefore concentrated at the large Maria Island reserve, where six sites in the reserve and six sites outside were monitored. Two sites were monitored both inside and outside the other reserves, except at Ninepin Point where the small region of reef included sufficient area for only one site inside the reserve to be assessed. The locations of these monitoring sites are shown in Fig. 1.

External reference sites were selected to encircle reserves at approximately equal-spaced distances, and as the best match of wave exposure and macroalgal communities to the reserve sites (Edgar and Barrett 1997). A consequence of this procedure was that the spatial separation of reserve sites (1-7 km) was less than the spatial separation of associated reference sites (3-20 km). Multivariate analyses using MDS indicated that plant and animal assemblages at reference sites always corresponded more closely with associated reserve sites than with sites at other reserves (Edgar and Barrett 1997).

Monitoring commenced at the Maria Island, Tinderbox and Ninepin Point marine reserves in March 1992 and at the Governor Island Marine Reserve in August 1992, with surveys undertaken each year in autumn until 2001 and in the spring of 1993, 1994 and 1997. The exception to this protocol was that no monitoring was conducted at Governor Island in 1995 or 1996.

Spatial variability between sites was reduced by only obtaining data along the 5 m (±1 m) depth contour, except on the deep reef at Governor Island where additional sites at 10 m inside and outside the reserve were included as the additional reef depth. Exposure at this location meant 5 m deep transects represented only a small proportion of the habitats present. The 5 m depth was considered optimal for monitoring because:

(i) few reefs in reserves other than Governor Island extended below 7 m,

(ii) shallower habitats were difficult to sample because of near-vertical slopes in some areas and wave turbulence,

(iii) diving times were not limited by decompression schedules, and

(iv) reefs at 5 m are subjected to heavy fishing pressure from net and rock lobster fishers and divers.

2.1.2 Census methodology

Visual census techniques were used in the study because sampling needed to be non-destructive within reserves and a large amount of data was required on a range of species within the short seasonal survey periods. Three different census methods were used to obtain adequate descriptive information on reef communities at different spatial scales. These methods are described briefly below and in more detail elsewhere (Edgar *et al.* 1996, Edgar and Barrett 1997).

At each reef site, the abundance and size structure of large fishes, the abundance of cryptic fishes and benthic invertebrates, and the percent cover of macroalgae were censused separately. The densities of large fishes were estimated by laying four 50 m transect lines along the 5 m depth contour. A diver swimming up one side of the line and then back along the other recorded the number and estimated size-class of fish within 5 m of each side of the line on waterproof paper.. Size-classes used in the study were 25, 50, 75, 100, 125, 150, 200, 250, 300, 350, 375, 400, 500, 625, 750, 875 and 1000+ mm. A total of four 5 m x 100 m transects was censused for large fish at each site. The distance between ends of adjacent transects was small (0-5 m) relative to the length of transects (50 m), consequently the four transects at each site were considered subsamples, which indicate variability within the site, rather than as true randomly distributed replicates.

Smaller fishes and megafaunal invertebrates (large molluscs, echinoderms, crustaceans) were next counted along the transect lines used for the fish survey by recording animals within 1 m of one side of the line (a total of four 1 m x 50 m transects). The distance of 1 m was assessed using a stick carried by the diver. The maximum length of abalone and the carapace length of rock lobsters were measured underwater using vernier callipers whenever possible.

The area covered by different macroalgal species was then quantified by placing a 0.25 m2 quadrat at 10 m intervals along the transect line and determining the percent cover of the various plant species. Cover was assessed by counting the number of times each species occurred directly under the 50 positions on the quadrat at which perpendicularly placed wires crossed each other (a total of 1.25 m2 for each of the 50 m sections of transect line).

In order to reduce variability in estimates attributable to different divers, all plant data were obtained by either Barrett or Edgar and approximately 60% of fish and invertebrate data were obtained by these researchers. The remaining data were obtained by several other divers, with a single additional diver used in each season for surveying fish at both reserve and reference sites. One additional trained diver assisted in invertebrate searches in 1999, 2000 and 2001. Biases associated with each diver should be almost evenly distributed between reserve and reference sites.

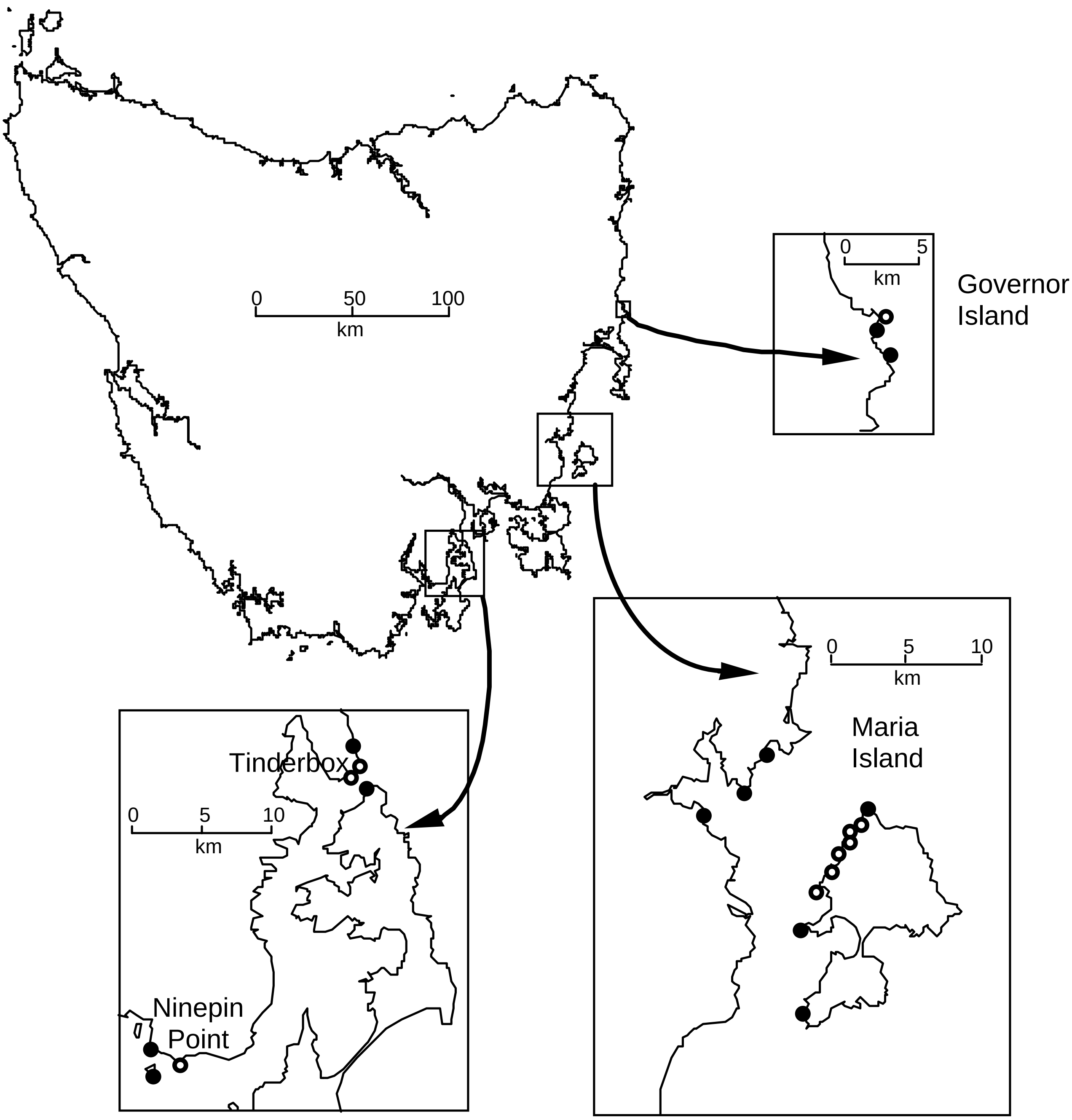

Fig. 1. Map showing distribution of study sites along the eastern Tasmanian coast. Sites with open circles were located within reserves, solid circles indicate external reference sites. At Governor Island at Bicheno the two reserve sites show as a single circle at this scale.

The data collected during this study over the period 1998 to 2001 was not analysed in detail because the project scope was to gather sufficient data to allow a detailed analysis of changes accumulating over a ten year period (1992-2002) if sufficient additional funding was obtained. The Tasmanian Aquaculture and Fisheries Institute has obtained FRDC funding (Project 1998/162) for this purpose and a detailed analysis will be done during 2002. Data presented in this report is therefore essentially summary data of the more noteworthy results obtained from the time series of monitoring at Maria Island. As such they are simply mean values and standard errors and no analysis has been conducted into the significance of the results.

2.2 Community Monitoring

2.2.1 Diver involvement

Over the three summer periods that community monitoring was conducted, project officers were appointed to enthuse and involve dive clubs and interested individuals, organise dives, prepare datasheets and interpretive material, enter and check the data obtained.

2.2.2 Diver training

Volunteers were trained in the methodology and in identification skills at the study sites. Before each dive a two hour briefing was given explaining the methodology and introducing the range of species that may be encountered at each location. This included familiarisation with all commonly encountered fishes and invertebrates at each location as well as familiarisation with the use of and identification of common species shown on detailed underwater identification sheets.

2.2.3 Locations surveyed

The survey locations were restricted to places that could be easily and safely shore-dived. These were at Ninepin Point Marine Reserve (one site, Ninepin Point central)(Fig. 1), Tinderbox Marine Reserve (3 sites, Tinderbox boatramp, Tinderbox middle site, Piersons Point)(Fig. 1) and at the Crayfish Point Marine Reserve at Taroona, a suburb of Hobart on the southern shore of the River Derwent. Ninepin Point and Tinderbox are fully protected reserves and the Crayfish Point reserve is partially protected. At the latter site, hand fishing is allowed while the use of recreational and commercial gillnets and the taking of fish, lobsters and abalone by divers is prohibited. In 1999 surveys were conducted at Ninepin Point, Tinderbox ramp, Tinderbox middle and Piersons Point. In 2000 surveys were conducted at Tinderbox middle site and Ninepin Point (x2). In 2001 surveys were conducted at Ninepin Point, Tinderbox middle site (x2), and at Crayfish Point. In all years a minimum of six weekend diving sessions were organised, however the number of dives that were successfully completed was restricted by weather and diver availability.

2.2.4 Volunteer methodology

Volunteer divers used essentially the same quantitative line transect methodology as that used by research staff in the long-term monitoring program detailed earlier in the methods section. To simplify the task volunteer surveys were restricted to fishes and invertebrates. At the range of locations surveyed, attempting to describe the algal diversity encountered would have substantially increased the complexity of the task asked of volunteers and of the project officer overseeing each community dive.

The most important component of dive based projects involving volunteers is diver safety both in and out of the water, so dive locations were carefully selected to minimise risk. To ensure diver safety, the methodology was modified to ensure all divers were in buddy pairs and in very close proximity to each other at all times. At each site to be surveyed, experienced divers swam the survey reels out to a central point at five metres depth. This position was noted for repeat surveys in the following years. Using a transect line wrapped around a wire hose reel, the volunteer divers swam and unwound a weighted 100 m transect line from the central point, swimming along the coast on the 5 m depth contour.

The volunteer divers thus conducted a 100 m x 5 m fish transect, swimming in buddy pairs from the central point, counting the abundance and estimating the length of each fish encountered. Each diver swam in the middle of a 5 m wide lane on one side of the transect. The buddy swam an identical lane on the other side of the transect, so divers were never more than 5 m apart. The middle of the lane was calibrated by the diver measuring this distance along the transect line which was marked at 50 cm intervals. For species of fish that school in large numbers (such as *Trachinops caudimaculatus)*, fish within each school were counted up to 20 individuals and then the area occupied by the block of 20 fish was used to estimate the total abundance of the school.

Upon completion of the fish transects, the divers each conducted a 1 m x 50 m invertebrate and cryptic fish search with the divers working side by side, one on each side of the line. The divers comprehensively searched the area from the line to a distance of one metre. This distance is estimated by divers carrying one metre long measuring sticks or using a known distance measured across their body. This protocol (one fish count and one invertebrate search) took approximately 40 minutes to complete, which is the average duration of a recreational dive. It allowed sufficient air to be left in tanks for a good safety margin, or for participants to have a more relaxed swim about the site before exiting the water. Where four volunteer divers participated the survey covered the same area that would normally be censused during the annual monitoring at each location.

2.2.5 Debriefing

An important part of each volunteer event was the post-dive debriefing session, usually held in conjunction with a BBQ. During this time the event coordinator discussed with the group all of the species encountered and helped with difficult identification. . The coordinator then ensured all divers had identified each species on their datasheets and that the date, location surveyed, position on transect censused and the name of participants was recorded. Another important component was to ensure that all hand writing on the sheets was legible as often underwater writing is particularly difficult to interpret.

2.2.6 Validation

To review the quality of data obtained from the volunteer divers concurrent dives were conducted at most of the sites surveyed by divers skilled in conducting underwater census work. The average results of these divers were then compared with the results from volunteer divers.

2.2.7 Analysis

The results of each dive were presented as counts by each diver at each location surveyed. This was with the exception of the second dive at Ninepin Point in 2000 where it appeared the datasheets were misplaced before the data was entered onto our database. Pooled total abundances were presented for each species at each location with the totals calculated from the mean value of each transect. If two or more divers completed the same transect the mean of their estimates was obtained and used in the calculation. In the unusual circumstance where only three of the four fish transects were completed, the total was multiplied by 4/3 to estimate the total if the whole site had been surveyed. With the invertebrate data, estimates within each 50m block were pooled to give a mean value for that block. Usually this involved one survey on the inside of the line and one survey on the outside of the line (by a buddy pair) but occasionally more than two searches were conducted if a large number of volunteers were present.

To investigate the variability between divers (community and skilled) the level of similarity between the fish and invertebrate abundance results of individual diver transects (at sites where large numbers of divers participated) were compared using the Bray-Curtis similarity index. This index calculates a pairwise matrix of similarity between all diver replicates. The abundance of observed species was used in the analysis with the data square root transformed before calculation of the index. Species known to be pelagic vagrants or uncommonly encountered schooling species were not included in the calculations. These included *Sardinops sp.* a pelagic species and *Dinolestes lewinii,* a schooling species often associated with reefs. The similarity matrix was agglomeratively clustered using ranked data and group-averaging as suggested by Clarke (1993) and graphically presented using multi-dimensional scaling (MDS). Analysis was performed using the PRIMER software developed by the Plymouth Marine Laboratories. The ability of volunteer-based average abundance data to detect stable ecological differences between survey sites for each site censused was investigated using the same methodology. The results are presented as a series of MDS plots for fishes and invertebrates (plus cryptic fishes) using the Tinderbox mid-site and Ninepin Point data over three years.

3. Results

3.1 Long –term monitoring

A detailed analysis of the results obtained during ten years of monitoring biotic communities within and adjacent to Tasmania's MPA's will be completed in 2002 following the completion of surveys in Autumn and Spring of that year. The following results are a preliminary examination of some of the results at, and adjacent to the Maria Island marine reserve after a nine year period of protection (1992-2001).

3.1.1 Lobsters

The mean size of lobsters has continued to change within the reserve due to protection from fishing (Fig. 2), increasing from 90 mm in 1992 to 116 mm in 2001, while the mean size at reference locations outside the reserve remained stable (86 mm in 1992 and 83 mm in 2001) and is now significantly less than the average size within the reserve.

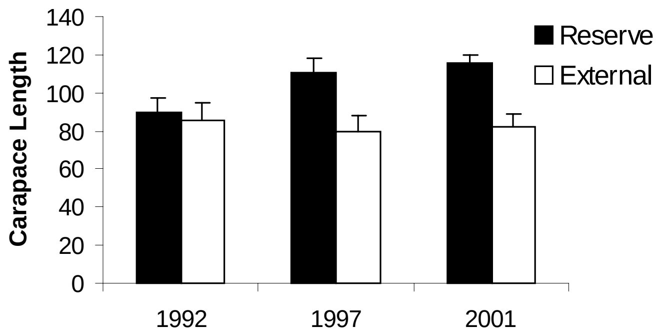

Fig. 2. Lobster (*Jasus edwardsii*) mean carapace length (with S.E.) for the Maria Island marine reserve and external reference sites. Mean is mean of site means (n = 6).

The total abundance of lobsters within the reserve appears to have stabilised and may now have reached an equilibrium between recruitment and mortality (Fig. 3). After increasing more than threefold from 0.79 to 2.8 lobsters per transect in the first five years of protection, there has been little change between 1997 and 2001. At the external reference sites the mean abundance has hardly changed since 1992, with mean abundances of 1.35, 1.52 and 1.50 respectively being recorded in the years 1992, 1997 and 2001.

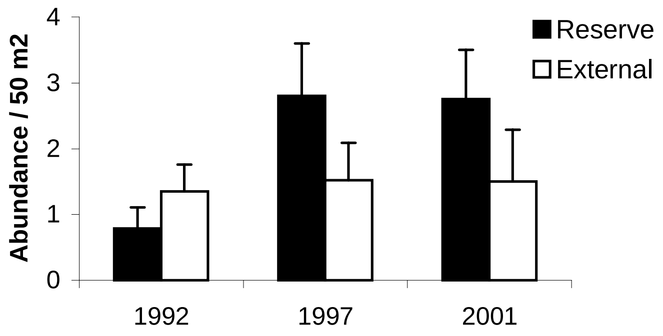

Fig. 3. Lobster (*Jasus edwardsii*) mean abundance per transect (with S.E.) for the Maria Island marine reserve and external reference sites. Mean is mean of site means (n = 6).

3.1.2 Abalone

The mean size of abalone within the reserve has slowly increased with time (Fig. 4), from 128mm in 1992 to 141mm in 2001. At the external reference sites the mean size has fluctuated from 125mm in 1992 to 118mm in 1997 to 123mm in 2002, a size significantly smaller than that found within the Maria Is. reserve. The abundance of abalone has declined markedly within the Maria Is. reserve (Fig. 5) between 1992 and 2001, with abundances falling by over 50% from 15.5 abalone per transect in 1991 to 7.4. At the same time the abundances at the external reference sites have remained relatively stable, fluctuating between 9.2 to 11.5 to 7.3 over the three years that mean values were calculated.

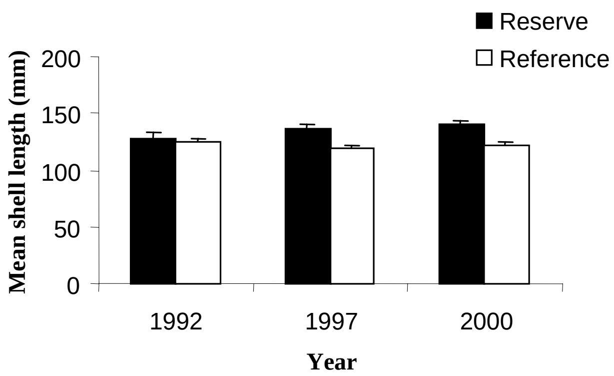

Fig. 4. Blacklip abalone (*Haliotis rubra*) mean shell length per transect (with S.E.) for the Maria Island marine reserve and external reference sites. Mean is mean of site means (n = 6).

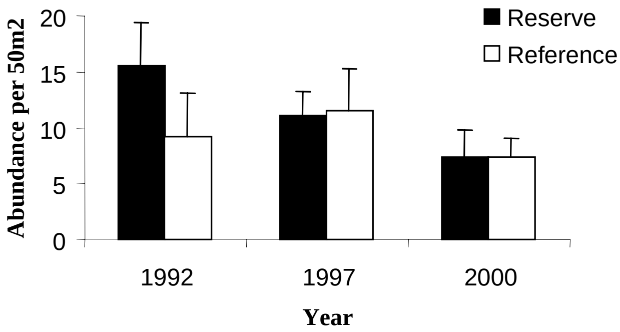

Fig. 5. Blacklip abalone (*Haliotis rubra*) mean abundance per transect (with S.E.) for the Maria Island marine reserve and external reference sites. Mean is mean of site means (n = 6).

3.1.3 Additional species

The common urchin *Heliocidaris erythrogramma* is the most dominant large invertebrate within the Maria Island reserve. The abundance of this species within the reserve appears to have declined substantially between 1992 and 2001 (Fig. 6), falling by approximately 40% from 106 to 67 urchins per transect. At the same time the external sites have experienced fluctuations, but the overall decline at these sites is minimal (16% from 135 to 113 per transect).

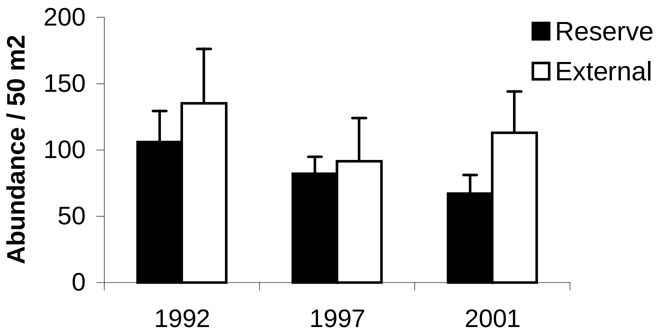

Fig. 6. Common urchin (*Heliocidaris erythrogramma*) mean abundance per transect (with S.E.) for the Maria Island marine reserve and external reference sites. Mean is mean of site means (n = 6).

One of the more notable features during the annual surveys has been the increasing numbers of the long-spine urchin *Centrostephanus rodgersii* within this region. The average abundance in the vicinity of Maria Island has increased by nearly five times during the period 1992 to 2001, showing a gradual progression through time (Fig. 7). Most of this increase comes from the more oceanic sites at Isle du Nord and Green Bluff at the northern and sothern ends of Maria Island, but this species is found in variable abundances at all sites surveyed.

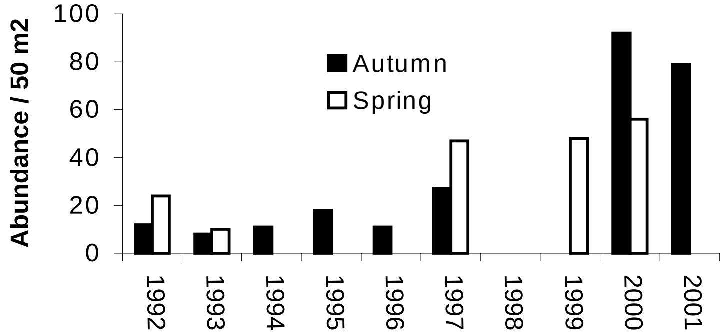

Fig. 7. Long-spine urchin (*Centrostephanus rodgersii*) total abundance recorded per year within the Maria Island marine reserve and at external reference sites.

The introduced brown algae *Undaria pinnatifida* is found throughout the Maria Island reserve and at all external reference locations. It is an annual species with a macroscopic sporophytic stage that usually develops from a small plant in June to a large plant of 1.5m length in late November. Plants become necrotic throughout December and usually are only present as residual holdfasts and sporophylls by March when the Autumn surveys are conducted. Figure 8 demonstrates this with moderate abundances being detected in Spring (mid to late September) in the years where spring surveys were conducted. Abundances are substantially less in autumn following the dieback, and abundances at this time of year are usually low but can ocasionally reach 0.5%. Interrannual variation in abundance was significant although there is no evidence of a trend for increasing dominance of the algal community by this species over the years 1992 to 2000.

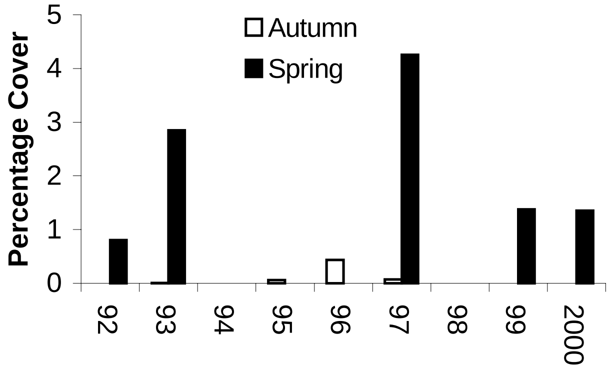

Fig. 8. Average percentage cover of the introduced brown algae *Undaria pinnatifida* recorded per year within the Maria Island marine reserve and at external reference sites. Spring surveys were conducted only in the years where abundance is shown (1992, 1993, 1997, 1999 & 2000). Annual surveys were conducted in all years.

3.2 Results - Community monitoring

3.2.1 Individual diver results

Community monitoring events were conducted regularly between 1999 and 2001 with the results of all events and individuals entered onto a database for analysis. Most individual site/diver results are presented in the appendices, however some examples are included here. Tables 1 and 2 give an indication of the extent of concordance between volunteer results and the results of skilled researchers completing the same transect. Table 1 shows the invertebrate and cryptic fish results of individual divers from a monitoring event at Ninepin Point in April 1999. For the most common species there usually is a reasonable agreement between volunteers and skilled observers with volunteer values often falling between 50-200% of skilled estimates. Occasionally common species were ignored by volunteers overwhelmed by the complexity of the task, an example of this is V1 and V9 who did not count the common featherstar *Commanthus trichoptera* at all, or V11 who reported very low numbers of the common urchin *Heliocidaris erythrogramma* despite it being a readily identifiable species and abundant at that location.

Occasionally the volunteer data was strongly biased towards larger and familiar species, such as the unusually high abalone *Haliotis rubra* count of V9. Often additional species not detected by skilled observers were recorded by volunteers and it appears that volunteers are attracted to, and count, “interesting” species outside of the search area. One of these appears to be the velvet seastar *Petrecia vernicina*, where counts by V20 and V13 were substantially greater than those of skilled searchers.

During fish surveys there appeared to be a reasonable degree of similarity between the results of volunteers and skilled observers (Table 2). This was particularly the case for large site attached species such as the blue throated wrasse (*Notolabrus tetricus*) where values were often quite similar. The species *Trachinops caudimaculatus* (blotch tailed trachinops) often occurs in large schools, presenting some difficulty in obtaining reliable counts. Some volunteers appeared to be quite good at estimating abundances of this species while others eg. V13 and V5 virtually ignored them, presumably because of the degree of difficulty involved. As many of the fishes involved are mobile and low in abundance, counts are variable in time (highlighted by the variability between days of skilled counts) and little interpretation of differences between the groups with respect to the less abundant species can be made. When the abundance counts across all species are compared pairwise between all divers and transects using the Bray-Curtis similarity index and this relationship is graphically presented by MDS, the volunteer results show substantially more variability between divers and transects than do the skilled results. The two groups plot as two distinct clusters using both the invertebrate data (Fig. 9) and the fish data (Fig. 10). This distinct clustering indicates there are substantial differences between the volunteer and skilled results across a range of species counts, and that this difference is usually greater than the between transect variation observed with the skilled divers. The overall community description by individual volunteers appears to be a poor representation of real abundances.

Table 1. Abundance of large mobile invertebrates and cryptic fishes recorded during a community monitoring survey at Ninepin Point in April 1999. Position 1 is the block 0-50 m from the central point, position 2 is the block from 50-100 m. Volunteers are indicated by the V prefix in the diver code.

Table 1 continued. Abundance of large mobile invertebrates and cryptic fishes recorded during a community monitoring survey at Ninepin Point in April 1999. Position 1 is the block 0-50 m from the central point, position 2 is the block from 50-100 m. Volunteers are indicated by the V prefix in the diver code.

Table 2. Abundance of fishes recorded during a community monitoring survey at Ninepin Point in April 1999. Volunteers are indicated by the V prefix in the diver code. Diver codes separated by a comma are results by skilled staff obtained on the same transect on the previous day.

Table 2 continued.

Fig. 9. MDS plot of relationship between invertebrate counts of skilled and volunteer observers at Ninepin Point in April 1999. Data is square root transformed and counts are compared pairwise using the Bray-Curtis similarity index. Stress is 0.08 indicating this 2D plot is a good representation of the relationship between observer data.

Fig. 10. MDS plot of relationship between fish counts of skilled and volunteer observers at Ninepin Point in April 1999. Data is square root transformed and counts are compared pairwise using the Bray-Curtis similarity index. Stress is 0.09 indicating this 2D plot is a good representation of the relationship between observer data.

3.2.2 Pooled results

To obtain an indication of how well the results of volunteer divers as a group could describe a site, the results from individual divers and transects at each site were pooled to give a site total. Where individual transects were replicated by a number of volunteers a mean value was calculated for that transect. The volunteer derived values were contrasted with those obtained by skilled observers at each site (Tables 3 to 8). For fishes at most sites (Tables 3 to 5) there was good agreement between the abundances recorded by both groups. This was most notable for large and common resident species such as *Notolabrus tetricus*, *Pictilabrus laticlavius* (senator wrasse) and *Caesioperca rasor* (barber perch) but was generally evident across a wide range of species. For the small schooling species *Trachinops caudimaculatus,* volunteer counts were usually within the range 50-200% of skilled counts, which is a remarkable result given the difficulty in counting small schooling species and their typical variability in time at any particular location. Some schooling species such as *Sardinops neopilchardus* (sardines) and *Dinolestes lewinii* (yellowfin pike) are particularly mobile and patchy in their distribution, and the variation between observer groups for these species is more a reflection on chance encounters rather than any difference between observer reliability. Generally the abundance of most common species varied little over the three years of the study indicating that reef fish communities are relatively stable over this time scale. Most discrepancies appear to be due to the presence or absence of schools of mobile species such as *Dinolestes and Sardinops,* or to the inclusion of patchy aggregations of newly recruited juveniles such as the common bullseye (*Pempheris multiradiatus*).

Table 3. Total abundance of fishes at Ninepin Point estimated by volunteer (V) and skilled (S) divers at Ninepin Point in 1999, 2000 and 2001.

Table 4. Total abundance of fishes at the Tinderbox mid-site estimated by volunteer (V) and skilled (S) divers in 1999, 2000 and 2001.

The similarity between the pooled estimated fish species abundances of the two groups was examined by calculating the pairwise relationship between values obtained by the groups at Ninepin Point and Tinderbox in 1999, 2000 and 2001 using the Bray-Curtis similarity index and viewing this relationship with 2-dimensional MDS (Fig. 11). This allows for the variation between volunteers and skilled observers to be examined in relation to variation between locations and between years at those locations. The spatial distribution within the MDS output (Fig. 11) clearly indicates that the pooled community values closely reflect the skilled values on most occasions and that in order of variance, location differences then interrannual differences appear to be greater than those between skilled and volunteer estimates. The Tinderbox values form a distinct grouping from the Ninepin Point values and years are generally more spatially distinct than the groups.

Comparison of site abundance estimates from the community and skilled invertebrate data indicates that there is a good degree of agreement between groups (Tables 6-8). At most locations there was strong agreement for the abundance of most of the large conspicuous invertebrates such as the common urchin (*Heliocidaris erythrogramma*), the pencil urchin (*Goniocidaris tubaria*) and the common featherstar (*Commanthus trichoptera*). Most community estimates of the abundance of these species ranged within 50-200% of the skilled estimates. One notable exception was the large discrepancy for *C. trichoptera* at Piersons Point in 1999 where surveys were conducted on different days. Positioning of the transect may explain much of the variation observed.

Table 5. Total abundance of fishes at Piersons Point (at Tinderbox), Tinderbox central (boatramp) and Crayfish Point (Taroona) estimated by volunteer (V) and skilled (S) divers during community monitoring events conducted in 1999 and 2001. * Denotes skilled values derived from annual monitoring at that location on the day previous to the community event.

A similar variation in the count of *Coscinasterias muricata* (eleven armed seastar) also occured at that time. Some species were locally common, including *Uniophora granifera* (zigzag star) and *Jasus edwardsii* (rock lobster) at Tinderbox (Fig. 7) and *Nectria oscillata* (ocellate seastar) and *Petrecia vernicina* (velvet star) at Ninepin Point. Estimates of the abundance of these species also showed good agreement between community and skilled groups. The results indicated that the abundance of most species was relatively stable over the three year period of the study, with the most notable exception being *Coscinasterias muricata* at Tinderbox. *Coscinasterias muricata* is not usually observed in abundances greater than 5 per site but was recorded in numbers up to 540 per site at the Tinderbox sites in 1999, and then at one per site in 2001, demonstrating the value of community monitoring in being able to detect unusual biological events within reef systems.

Fig. 11. 2-dimensional MDS plot of the relationship between volunteer and skilled estimates of fish community composition at the Tinderbox mid site and Ninepin Point during surveys in 1999, 2000 and 2001. Based on pairwise comparison between groups using the Bray-Curtis similarity index on square root transformed abundance data, excluding *Dinolestes* and *Sardinops*. Stress is 0.06 indicating this 2-dimensional plot is a good representation of the relationship between the points. Symbols are Ti = Tinderbox, Ni = Ninepin Point, V = volunteer data, S = skilled data.

Table 6. Total abundance of mobile megafaunal invertebrates and cryptic fishes at Ninepin Point estimated by volunteer (V) and skilled (S) divers in 1999, 2000 and 2001. * Denotes values obtained during annual surveys of the site, conducted one week prior to the community survey.

Table 7. Total abundance of mobile megafaunal invertebrates and cryptic fishes at Tinderbox mid-site estimated by volunteer (V) and community (C) divers in 1999, 2000 and 2001. * Denotes values obtained during annual surveys of the site, conducted one week prior to the community survey.

The similarity between the pooled estimated invertebrate and cryptic fish species abundances of the two groups was examined by calculating the pairwise relationship between values obtained by the groups at Ninepin Point and Tinderbox in 1999, 2000 and 2001 using the Bray-Curtis similarity index and viewing this relationship with 2-dimensional MDS. The spatial distribution within the MDS output (Fig. 12) was similar to that found for fish counts. This clearly indicates that the pooled community values closely reflect the skilled values on most occasions and that in order of variance, location differences then interrannual differences appear to be greater than those between skilled and volunteer estimates.

Table 8. Total abundance of mobile megafaunal invertebrates and cryptic fishes at Tinderbox central (boatramp), Piersons Point (at Tinderbox) and Crayfish Point (Taroona) estimated by community (C) and skilled (S) divers in 1999, 2000 and 2001. * Denotes values obtained during annual surveys of the site, conducted one day prior to the community survey.

The Tinderbox values form a distinct grouping from the Ninepin Point values and years are generally more spatially distinct than the groups. There was greater distance between the 2001 community and skilled values than the average relationship and this appears to be related to volunteers detecting more species than the skilled group. In 2001 the two surveys were conducted on different days adding to variation in the comparison. The increased level of species detected by the volunteers at some locations may often be an artefact of obtaining a mean value where several volunteers repeated the same transect but saw a different set of species, however at this location there was no replication of volunteer transects. The additional species count appears to be related to a bias of volunteers towards counting “attractive” species that lie in their field of vision but outside the one metre wide search area.

Fig. 12. Two-dimensional MDS plot of the relationship between volunteer and skilled estimates of large megaufaunal invertebrate and cryptic fish community composition at the Tinderbox mid site (T) and Ninepin Point (N) during surveys in 1999 (99), 2000 (00) and 2001 (01). Counts of featherstar (*Commanthus*) species and short-spined urchin species (*Amblypneustes* and *Holopneustes*) are pooled as volunteer were not trained to recognise the difference between very similar species. Sites Ninepin 2000 community, Ninepin 2000 skilled and Ninepin 2001 skilled closely overlap at the mid left edge of the plot. Stress is 0.05.

3.2.3 Size estimation

During the community monitoring dives volunteers were asked to estimate the size of fishes observed using lengths marked on their underwater slates as a size reference. For many species the abundance at each site was too low to obtain meaningful estimates of site means for comparison between the two groups of observers. Two species that did have moderate abundances on a regular basis were bastard trumpeter (*Latridopsis forsteri*) (Fig. 13) and blue throated wrasse (*Notolabrus tetricus*) (Fig. 14). Comparison of trumpeter size estimates between volunteers and skilled observers at Tinderbox indicated that the volunteers consistantly underestimated the mean length and that this difference was approximately 25%. For the bluethroated wrasse, size estimates across a range of sites and times indicated that volunteer size estimates were usually substantially less than the skilled observers although the difference was usually significant, ranging up to 30% at Ninepin Point in 1999.

Fig. 13. Mean length (S.E.) estimate of *Latridopsis forsteri* at Tinderbox mid-site during three monitoring events. Values are calculated as the mean of all length estimates at each site at each time. Total number of fish in mean calculations is shown within each column.

Fig. 14. Mean length (S.E.) estimate of *Notolabrus tetricus* at Tinderbox mid-site, Ninepin Point and Crayfish Point over a number of monitoring events. Values are calculated as the mean of all length estimates at each site at each time. Total number of fish in mean calculations is shown above each column.

A summary of the estimated abundances of common invertebrate and fish species at Ninepin Point and Tinderbox sites observed by volunteer monitoring between 1999 and 2001 is shown in Figs 15 and 16. For easily recognised and long-lived invertebrate species such as *Heliocidaris erythrogramma*, *Goniocidaris tubaria* and *Petrecia vernicina* counts appear relatively stable through time, indicating not only a stability in the population of these species but also that they may be reliable indicator species for monitoring by volunteers. *Commanthus trichoptera* counts were less reliable due to the difficulty in counting this species. Individuals are often found in aggregations and without experience it is often difficult to determine abundances reliably. Several divers were overwhelmed by the complexity of this task and chose to ignore this species. One conspicuous species that was counted in large numbers at Tinderbox in 1999 was *Coscinasterias muricata* (Fig. 16). The counts in the following two years were more than two orders of magnitude less than in 1999, and regardless of the degree of variability of volunteer data this change in abundance is highly significant and biologically meaningful, demonstrating that volunteer data can be used to detect significant change. Most of the individuals counted were small, suggesting that there had been a strong recruitment pulse in that area in that year, and that they had either dispersed or died before the next monitoring event.

The variability in volunteer estimates of fish abundances depended on the species counted (Figs. 15 &16). For common long-lived reef residents such as *Notolabrus tetricus, Pictilabrus laticlavius* and *Caesioperca rasor* counts were relatively constant over the three years of monitoring, indicating a degree of population stability in these species. At Ninepin Point the abundance of another common reef resident *Notolabris fucicola* declined markedly over this period. This decline was mirrored by similar results from the skilled estimates (Table 3), indicating that this was a significant trend and that the results of volunteer monitoring were sufficient to detect this decline. Schooling species such as *Trachinops caudimaculatus* had highly variable abundances and this variability was mirrored in the skilled estimates. While *Trachinops* has been variable, other schooling species have shown distinct trends. A recruitment of bastard trumpeter (*Latridopsis forsteri*) onto the Tinderbox reef led to a small population residing there between 2000 and 2001, while a chance occurrence with a passing school of jackass morwong (*Nemadactylus macropterus*) at Tinderbox caused a temporary peak there.

Fig. 15. Examples of change in estimated abundances of common invertebrates during three years of community monitoring at the Ninepin Point Marine Reserve.

Fig. 16. Examples of change in estimated abundances of common invertebrates during three years of community monitoring at the Tinderbox Marine Reserve.

4. Discussion

4.1 Long-term monitoring

Long-term studies of the patterns and processes occuring in our coastal marine environments are essential if we wish to understand and manage our influence on coastal species and ecosystems. The results of a study examining changes occuring in the first five years of protection of coastal marine systems in no-take marine reserves in Tasmania (Edgar and Barrett 1997, Barrett and Edgar 1998, Edgar and Barrett 1999) demonstrated that fishing had substantially altered the population demographics of a number of commercially important species, and had resulted in a reduction in the level of biodiversity when measured at the scale of an individual location. While few other studies have documented change through time, there are now numerous publications demonstrating differences in the abundance of fished species between protected and unprotected areas. These have been reviewed by Roberts and Polunin (1991), Marine Reserves Task Group (1997), Ward *et al.* (2001) and Roberts (2001) and clearly show that MPA's provide protection from fishing and other human activities, and that this protection can lead to substantial change in the marine biota. The magnitude of the difference between protected and unprotected areas will obviously depend on the degree to which fisheries are managed in areas outside the reserve. There is considerable community interest in determining the extent of differences between protected and unprotected areas and in gaining information on the current level of fishing impacts. In an ideal situation, with well managed fisheries, the abundance and average size of commercially targetted species between fished and protected areas should not be too great as decreases below 50% of the virgin stock levels of many species may lead to an overall reduction in the productivity of the fishery (Cushing 1975). Unfortunately ideal situations rarely exist and often fisheries are managed in the absence of any data on current verses historical stock structures. Even with good management of the target species there may still be substantial impacts on bycatch species, the statistics of which are rarely collected and documented. One of the greatest benefits of MPAs is that they can provide this information for both target and bycatch species if these are resident species.This allows us to discuss and determine acceptable levels of fishing and its wider influence on species diversity through bycatch and cascading ecosystem effects.

The current results from monitoring at Maria Island indicate that for one heavily fished species, rock lobster, there has been a substantial recovery in population numbers within the reserve in the nine years following protection. Total numbers increased more than threefold and the biomass of mature individuals increasing more than fivefold. It appears that the population increase documented after the first five years of protection (Edgar and Barrett 1999) has now slowed and that the population may have reached an equilibrial state. If this is the case we will now have sufficient information to determine the direct impact of this fishery on lobster populations within this region and to facillitate discussion on the appropriate stock levels to be maintained on a regional basis for this species to managed for optimal productivity.

While we are now close to understanding the direct impact and magnitude of lobster fishing, there are potentially cascading ecosystem effects caused by removing large numbers of an important predator in Tasmanian inshore reef ecosystems. The substantial decline in urchin numbers within the Maria reserve over the nine years following protection (40%) may be strongly related to the increase in abundance and size of lobsters, as urchins are known to be an important component of lobster diets (Edmunds 1996, Frusher and Edmunds 2000) and lobsters have been documented to play an important role in regulating urchin numbers (eg Baraki and Branch 1991, McClannan and Shafir 1990, Tarr *et al.* 1996, Anderson *et al.* 1997) on both temperate and tropical reefs. Large fish predators can also influence urchin numbers, and it is possible that the decline in urchin numbers is a result of a general increase in the abundance and range of urchin predators. Untangling this relationship would benefit from exclusion experiments to determine the relative influence of lobsters and fishes on urchin densities, and preliminary results from a PhD study within the Maria reserve indicate that large lobsters have the ability to prey on urchins of all sizes (Hugh Pedderson, School of Zoology, University of Tasmania pers. comm.). In northern New Zealand, lobsters and snappers have been identified as important factors in the control of urchin density (Cole and Keuskamp 1998), with urchin dominated substratum now occupying considerably less reef in reserves than in unprotected areas (14% cw 40%), and with urchin densities declining from 4.9 to 1.4 per m^2 between 1978 and 1998. Now that the lobster population at Maria Island contains a substantial proportion of large individuals capable of preying on the biggest of urchins, a similar decline in urchin numbers could be possible over the next decade.

A recent review of literature on the ecosystem effects of fishing in kelp forest communities (Tegner and Dayton, 2000) indicated that one of the greatest impacts related to fishing was the destructive overgrazing of macroalgae by sea urchins. When their main predators are removed by overfishing, urchin densities can increase to such an extent that they substantially alter their surroundings. In this sense "overfishing" is the removal of key species to the extent that substantial ecosystem shifts occur, rather than to a level where productivity of the fishery is less than optimal. Once destructive overgrazing occurs there are cascading ecosystem effects, as other species dependent on the kelp for food and habitat become adversely effected. In Tasmania, the extent that urchin densities may have changed in response to fishing pressure is not yet clear. While urchin barrens are present at some locations, they are not common and the effects of urchin grazing may be more subtle. Ongoing monitoring of the current marine reserves and their associated fished reference sites will play an important role in understanding the mechanisms and magnitude of this process over the next decade, as this is the time frame that gradual changes in response to increasing predator sizes and abundances may become evident.

Blacklip abalone is another important commercial species in Tasmanian waters and may be threatened with local overfishing especially in some areas of the east coast (Tarbath *et al.* 2001). Changes in the abundance and size distribution of abalone have been closely monitored at Maria Island. After nine years of protection it appears that the mean size within the reserve has increased by approximately 15%, providing evidence that fishing for abalone has altered the population structure of this species in the study area and potentially more widely throughout eastern Tasmanian waters. This appears to be relatively stable at present, as the mean size at reference sites has shown no evidence of continuing decline or recovery over the nine year period of the study.

Unlike lobster numbers, mean abalone abundance within the Maria Island reserve appears to have declined substantially over the period between 1992 and 2000 regardless of the protection from fishing. While this decline was not statistically significant when examined after five years of protection due to a high level of variance in abundance estimates (Edgar and Barrett 1999), it now appears as a significant trend. Abundance estimates at the reference locations show no significant change. There are a number of possible reasons for a decline in abalone numbers within the reserve, including intraspecific competition, increased predation and delayed emergence. Manipulative experiments will be needed to identify the main factors involved. Certainly protection from fishing within the reserve has lead to a significant increase in the abundance and mean size of abalone predators (Edgar and Barrett 1999), including wrasse and lobsters, species known to be important abalone predators (McShane 2000, Shepherd 2000). Lobsters have been documented to play a pivotal role in structuring invertebrate assemblages on reefs in South Africa (Barkai and Branch 1988) and it is possible they could perform a similar role here, when present at natural levels of abundance.

When the size distribution of abalone within the Maria Island reserve was examined after five years of protection, the density of small abalone (less than 145 mm shell length) within the reserve was found to have significantly decreased, while the density of large abalone had significantly increased (Edgar and Barrett 1999). This trend appears to have continued, with the autumn 2000 results showing a slight increase in numbers in the largest size categories, and an overall decrease in the smaller size categories. The factors responsible for the decline in abalone abundance within the Maria Island reserve therefore appear to be acting predominantly on smaller abalone. If increased levels of predation are responsible for density reductions, this is the pattern that may be expected, with the vulnerable small size classes undergoing the most substantial decline. Alternatively a similar pattern may be produced if the presence of increased numbers of large individuals delays emergence, or if interspecific competition leads to decreased survival of small animals, particularly if the competition is for space. Ultimately a series of manipulative experiments will be required to determine the main causative factor(s).

One of the roles of long-term monitoring is to describe and increase our understanding of long-term environmental cycles and change. An almost complete lack of baseline data other than presence/absence information from museum records means that our interpretation of any change in the abundance or distribution of our coastal biota is particularly limited. The study at Maria Island has shown that many species appear to be relatively stable components of the marine community over the time frame examined so far. One of the notable exceptions is the long spine urchin *Centrostephanus rodgersii*, a species that has recently undergone a substantial range extension as a result of an increase in the extent of warm water intrusions onto the Tasmanian east coast. The warm waters associated with the east Australian current transport the larvae of a range of typically more northern species southwards during the summer period. Juveniles of a number of vagrant northern species are regularly encountered along the Tasmanian east coast during late summer and into autumn but rarely survive the cold winter. With a CSIRO monitoring station recording an increase in average water temperatures off Maria Island of between 1 and 1.5 degrees Celsius over the past 30 years (Edgar 2000) some of the vagrant species are undergoing range extensions. The five fold increase in

Centrostephanus abundance at the Maria Island sites is evidence of at least one species surviving multiple winters to become a conspicuous component of the reef fauna.

Another species undergoing a range extension in the vicinity of Maria Island is the introduced algae *Undaria pinnatifida.* This algae was probably introduced to Tasmanian waters in the 1980's by woodchip vessels discharging ballast waters at the port of Triabunna on the coastline adjacent to Maria Island (Sanderson and Barrett 1988). Since then has increased its range as far north as Bicheno and as far south as Tinderbox where it is now seasonally abundant in the central section of the Tinderbox marine reserve. One of the core benefits of long-term monitoring is being able to document and describe changes in community structure as a result of invasion by exotic species, and use this information to predict community responses at other locations. At Maria Island the abundance of *Undaria* appears to have been relatively stable over the past nine years, undergoing fluctuations but not showing a strong increasing trend. These preliminary observations suggest that this species has reached its capacity to invade these habitats under current conditions. Individual plants are most common in disturbed areas around the lower reef fringe or in the immediate subtidal zone, or in areas that have been recently disturbed by storms or other physical processes. Recent results from an urchin barren zone within the Tinderbox reserve indicate that the high level of disturbance caused by urchin overgrazing is sufficient to allow a major invasion by *Undaria,* with this species recording 100% cover at depths of 3 to 4 metres within areas that would be normally defined as barrens. As the reserves reach a natural balance between predators and grazers, and algal communities respond to decreased levels of grazing, the algal community may become more resilient to invasion. By examining changes over the next decade we should be able to gain some insight into the ability of natural areas to withstand invasion by exotic species and the extent that disturbance of the remaining areas might make them susceptible to invasion.

Introduced species clearly present problems for MPA's where their intended role is nature conservation and the maintenance of biodiversity. As successful introduced species are by their very nature highly invasive, they are difficult if not impossible to exclude from MPA's, and must therefore have some impact on biodiversity at the local scale within these areas. Long-term monitoring of biodiversity levels within and adjacent to MPA's will help quantify the impact of invasive introduced species on local biodiversity and perhaps more importantly, the degree that disturbance related to fishing and other human impacts plays in facillitating the invasive process and the extent that natural assemblages within MPA's can be altered by introduced pest species. .

The changes presented here are a small subset of the data that will be analysed in detail in 2002 following a decade of protection of Tasmanian marine reserves. They do, however, illustrate that the reserves work effectively in protecting some species from the effects of fishing and can therefore conserve biodiversity at the local (reserve) scale. They can provide an indication of the magnitude of changes caused by fishing on the target species, and in the longer term, help us understand the system-wide effects of fishing. This information is of substantial value for ensuring fisheries are managed sustainably, and that the wider ecosystem effects of fishing are understood and managed for the conservation of biodiversity and the systems of which they are an important part. The long-term monitoring has revealed patterns associated with long-term climatic cycles and the invasion of exotic species. With a combination of ongoing monitoring over the next decade and targeted manipulative experiments investigating the patterns revealed by monitoring, Tasmania's marine reserves will play an important role in developing our understanding of coastal ecosystems, of the ecosystem effects of fishing, of the design and management of MPAs and of ways to better manage our fisheries.

4.2 Community Monitoring

4.2.1 Multivariate description of "place"

As a multivariate description of community structure, the results obtained from volunteer surveys of reef biota closely approximated those obtained by skilled observers. Differences at the level of "site" and "time" were clearly differentiated, with volunteer data usually explaining temporal variation within each site in a similar manner to skilled observer data, suggesting data collected at this level may be adequate in defining a particular place at a particular point in time. Schmitt and Sullivan (1996) found a similar pattern with trained volunteers in the Florida keys, where fish abundances recorded in log 10 categories at a range of sites produced spatial patterns in species abundance that were similar to those obtained by experienced scientists during previous studies of the Florida Keys. Our results suggest that volunteer monitoring, if adequate quality control is maintained, can yield valuable information on spatial and temporal patterns of variation within reef ecosystems. If organised on a large scale, surveys of this type could provide substantially more information than is currently available or afforded by research organisations. This information could then be used to generate questions and hypothesis for targeted research by experienced researchers, and where significant change is detected, flag the need for a detailed examination of the causes of this change, particularly if it appears to be anthropogenic.

One aim of this study was to examine if the accuracy and precision of the volunteer data improved with volunteer training and experience in conducting surveys.

Unfortunately volunteer interest was difficult to maintain through time and insufficient volunteers regularly attended monitoring events to be able to examine any trend. Studies on coral reefs have obtained mixed results with Dartwall and Dulvy (1996) and Millar and De'ath (1996) finding divers do improve accuracy and precision with training, while Musso and Inglis (1998) and Mumby *et al.* (1995) found no clear evidence of improved accuracy with training. They did however, detect notable differences with diver experience (hours logged). Given the opportunity to obtain feedback on factors such as species identification and methodology following each dive, it would be surprising if some improvement in volunteer results did not occur, at least over the period of the first 2-3 dives.

Common problems encountered during our monitoring events included:

- failure to complete transects due to confusion about the task,
- failure to count difficult species,
- failure to distinguish similar species,
- wildly inaccurate length estimates,

- failure to properly document the results, and
- aborted dives due to a range of difficulties related to diver experience.

Most of these problems could be overcome with additional monitoring experience, and individually focussed training and feedback to help overcome identified difficulties. As an example, some divers failed to count common and abundant species on their first dive, but following a debriefing session after the dive, were aware of the species involved and the importance of counting them on subsequent dives. The array of species to be identified in temperate waters is significantly less than those encountered in the tropics where current assessments of the reliability of volunteer monitoring have been conducted (eg. Musso and Inglis 1998, Dartwall and Dulvy 1996), and the mix of less species and more easily distinguished species would probably contribute to an improved learning ability. It is highly likely that if divers were trained in the methodology over a period of several days before field data were recorded (eg. Dartwall and Dulvy 1996, Mumby *et al.* 1995) the volunteer data obtained during this study would have been substantially more accurate than currently obtained.

Pooled data obtained from volunteers at each location and time was remarkably similar to that obtained by skilled observers. Data from individual divers indicated there was substantially greater variation between the results of volunteer divers than the results of the skilled divers, indicating that individual volunteer data may often be quite unreliable. While this variability would almost certainly been reduced following a period of in-field training and feedback, it indicates the value of pooling data at each site from monitoring events and of having replicate data for quality control, rather than relying on individual data. Musso and Inglis (1998) identified considerable within pair variability between volunteer divers in percentage cover of coral estimates, and suggested that replicates by additional divers may be a good idea for quality checking of individual data. Future volunteer monitoring programs should incorporate these observations into their experimental designs wherever possible.

4.2.2 Individual species abundances

In a similar manner to the multivariate data, the estimation of total abundance of individual species differed substantially between volunteers, while the pooled data from all volunteers from each site often closely approximated that obtained by skilled researchers. It is possible that with additional training, abundance estimates obtained by individual volunteers may eventually approximate those obtained by researchers. Estimates of the abundance of resident long-lived and common species such as the bluethroated wrasse (*Notolabrus tetricus*) were often very similar between the two groups of observers. With additional replication to balance out the increased variance of the data obtained during volunteer surveys, volunteers have the capacity to match skilled researchers in detecting biologically significant changes in the abundance of a range of common species, or species that may become common (eg. introduced pests). The types of change that may eventually be detected by reliable volunteer programs may include those associated with the declaration of MPA’s eg. (Edgar and Barrett 1999) or may be related to long-term environmental cycles and introduced species.

To be effective, the monitoring protocol used during this evaluation may need some modification as it relies on single replicates of a number of spatially related transects to adequately describe a location at a point in time. With volunteer surveys, multiple replicates of individual transects are needed for quality control. The current methodology requires eight volunteers so that each of the four transects is repeated at least once. Ideally more replicates are required to even out individual differences, and to obtain sufficient replication for examination of the diver related variance at each site and each transect. Modification of the methodology to reduce the number of transects to two would increase replication and ensure at least two replicates were available per transect when a group of four divers was used. Group sizes larger than this are often difficult to organise and then coordinate in the field, and the methodology needs to reflect this.

The reliability of abundance estimates for fishes or invertebrates obtained during studies of community monitoring programs on coral reefs were not comparable with this study. Coral reef studies usually examined abundance data at the family level (due to the excessive number of species to be identified) and were recorded either on a log 10 scale (Schmitt and Sullivan 1996), or in order of relative abundance (Mumby *et al.* 1995). Where total abundances were recorded, the data was not examined on a species basis (Dartwall and Dulvy 1996, Halusky *et al.* 1994). While not directly comparable, Musso and Inglis (1998) examined the ability of volunteers to estimate the percentage cover of structurally distinct groups of corals. They found that with limited training, volunteers could collect reliable quantitative information as long as the volunteers were experienced divers and had greater than 30 hours diving experience. This observation is equally likely to apply to abundance estimates of individual fishes and invertebrates by volunteers in the temperate zone. While no record was kept of diver experience during our study, we did find inexperienced divers often failed to complete transects, and regularly became confused about the task asked of them.

One definite outcome from the volunteer abundance data was that this information can be used to detect large changes in the abundance of species of interest. At Tinderbox this information enabled the detection of a strong pulse in the recruitment of the predatory, eleven armed seastar *Coscinasterias muricata* (a species known to play a key role in regulating scallop abundance). It also enabled the detection of a significant increase over the three year period in the abundance on the bastard trumpeter (*Latridopsis forsteri*), a popular target species of recreational fishermen. Annual monitoring by volunteers using the current methodology would be sufficient to detect most order of magnitude changes in the abundance of common species, and to detect the presence of unusual (including introduced) species at biologically low levels. In the absence of more rigorous data from locations such as this, abundance data on individual species derived from volunteer surveys can play a significant role in detecting large scale changes in population structure of key species. Such changes could be the focus of more intensive investigation by research organisations. One recognised benefit of volunteer monitoring is the ability to conduct surveys of individual target species rather than broadscale multispecies surveys. This ability is commonly used by researchers to record the abundance of a number of temperate species, ranging from introduced pests such as the north Pacific seastar (*Asterias amurensis*) to rare species such as the spotted handfish (*Brachionichthys hirsutus*).

4.2.3 Length Estimation

Length estimates for individual species could only be compared for *Notolabrus tetricus* and *Latridopsis forsteri* as these were the only species with sufficient abundance at either location for direct comparison of means. For both these species the lengths estimated by volunteers were highly variable when compared to skilled observers, with the estimate of mean size varying by up to 30% between the groups. This often appeared to be due to extremely high or low estimates by a limited number of volunteers, a factor than could probably be rectified by feedback and additional training and experience. Dartwall and Dulvy (1996) found that size estimates obtained by volunteers in Tanzania improved over three trials using model fish, to a stage where the frequency distribution obtained closely approximated the true value. With a similar level of training and feedback, community based volunteers in temperate Australia should be able to achieve similar results.

4.2.4 Logistics

Facilitating a community based monitoring program on temperate coastal reefs requires a substantial organisational input at a professional level. Many of these steps are described in detail in Musso and Inglis (1998) in their detailed assessment of the potential for developing reliable monitoring programs for coral reefs in Queensland. Essential components include appropriate training material and training methodology, underwater identification sheets and slates for recording observations, a simple, safe and workable methodology that will also yield meaningful results, an experienced facilitator and a group of enthusiastic volunteers. Additional components include finding safe and informative dive sites, providing clear briefings on the task, setting up dive transects, debriefing divers - including assistance in identification of unknown or mis-identified species, feedback on problems, ensuring sheets are legible and identification information is correct (eg. diver, site, date, position on transect, was the dive completed or aborted prematurely?), entry of data onto a database, and examination and presentation of results.

The greatest logistic difficulty encountered during this study was in organising sufficient (but not too many) volunteers to run a monitoring event, and having appropriate weather at the chosen location. Often events were cancelled because of bad weather, and this was complicated by the need to contact all volunteers to notify them of the change. Alternative site choice on a particular day was often impossible due to the inflexibility of coordinating eight independent volunteers. While there were many keen divers that wished to be involved in the monitoring program, participation on any particular day was limited by a number of factors. The key participating dive clubs were involved in a number of community participation programs including surveys for introduced species, removing introduced species from MPAs, cleaning up the River Derwent, conducting surveys for spotted handfish, replanting giant string kelp (*Macrocystis*) and surveying the abundance of scallops for a potential recreational fishery. These programs were in addition to the regular club dives. University students were the most enthusiastic group of divers but their free time was limited by the need to study. Our intention to examine the improvement of individual divers through time was limited by the fact that most divers only participated for one to two sessions. This was related to a mix of limited free time and that most participants used the monitoring program as a lesson in marine biology where they could learn some reef ecology, survey methodology and the names of reef species. After several sessions most participants had learnt enough and moved on. The colder waters of winter and spring further limited our ability to keep a regular program underway throughout the year, as interest quickly waned after May when water and air temperatures fell below comfortable levels.

4.2.5 Effectiveness of community monitoring

Volunteer monitoring programs have the capacity to provide large amounts of cost effective and reliable data under optimal circumstances. Our results indicate that at a low level of training, relatively inexperienced volunteers can provide sufficient information to adequately describe the biological assemblage at a particular location at a point in time. The reliability of this information decreases at the individual diver level or where individual species abundances are recorded or mean sizes are estimated. Studies of volunteer programs on coral reefs indicate that with sufficient training, using volunteers that are experienced divers, the results obtained can closely approximate those obtained by skilled researchers (eg. Dartwall and Dulvy 1996). This result is not surprising as the only difference between the two groups is the level of training and experience. The ultimate example of this is the Californian kelp monitoring program (Davis *et al.* 1988), a program that has established sufficient credibility to attract professional marine biologists as the volunteers.

At a small scale such as that undertaken in this study, the information collected by community based monitoring is likely to be more costly and less reliable than information collected professionally. To ensure a community based program is organised and executed properly, with sufficient quality control to ensure the results are considered reliable by all intended recipients of the data, a skilled facilitator needs to be employed to run the program. The costs of employing a person for one day to train volunteers, to organise each event, be present on the day to ensure adequate instructions, feedback and quality control is given, and to spend a day entering difficult data, is equivalent in cost to a professional dive team of three completing surveys at two or three locations. With the inflexibility of community events, time lost to bad weather or low visibility, and the low reliability of data requiring more replicates and monitoring events to obtain the same outcome. Skilled workers would complete approximately four surveys of differing locations at the same cost of one survey conducted by volunteers. This purely economic analysis ignores the obvious benefits of education and involvement that such events transfer to the wider community, but it does highlight the decisions that need to be made by funding agencies and the wider community when deciding on the most effective way of monitoring biological change through time. One clear exception to this is where volunteers are used to search for a particular target species that is readily identified and where large numbers of volunteer divers can be employed in the search. This methodology is often very cost effective if well coordinated, and provides benefits at all levels.

The optimal outcome of community involvement in the monitoring of reef biota would be to encourage sufficient interest to establish a strong volunteer organisation for coastal monitoring. The organisation would need a core of skilled volunteers that could train new recruits to a high level of proficiency before undertaking monitoring. It would also need a level of quality control that could convince professional agencies the data obtained were reliable and useful. There is sufficient evidence to show that data of this quality can be obtained by volunteers with adequate training. However, the greatest impediment may be putting this infrastructure in place and in developing and maintaining interest and momentum. Ultimately a partnership between community, research and management organisations has the potential to provide a cost effective method of monitoring reef ecosystems, with reliable community based monitoring at a comprehensive network of locations describing spatial and temporal patterns and processes and detecting significant change. Research organisations could then target studies on the major issues and processes identified, with management agencies responding to the outcomes. Until this goal is reached there has to be a recognition that long-term studies of processes occurring on temperate reefs are more reliably and economically conducted by trained observers, and if we are to develop an understanding of coastal processes and our impacts on them, research organisations will need to be funded accordingly.

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7. Appendix – Site abundance data by diver.

Table X1. Total abundance of cryptic fishes and invertebrates recorded during volunteer surveys at Ninepin Point in autumn 2000 and 2001. For position x = 0-100.

Table X2. Total abundance of fish recorded during volunteer surveys at Ninepin Point in autumn 2000.

Table X3. Total abundance of fish recorded during volunteer surveys at Ninepin Point in autumn 2001.

Table X4. Total abundance of cryptic fish and invertebrates recorded during volunteer surveys at Crayfish Point in autumn 2001.

Transect	Inner East	Inner West			Outer East			Outer West
Position	?	1	2	2	2	2	?	2
Species/Diver	V25	V20	V20	v22	V28	V28	V36	V21
Cryptic fish								
Cephaloscyllium laticeps	0	0	0	0	0	0	1	0
Invertebrates								
Echinoderms								
Comanthus trichoptera	3	5	1	2	0	2	0	1
Comanthus tasmaniae	0	1	0	0	0	0	0	0
Heliocidaris erythrogramma	28	5	26	2	5	4	10	3
Amblypneustes spp.	1	0	3	0	0	0	1	0
Coscinasterias muricata	0	0	1	0	0	0	0	0
Molluscs								
Turbo undulatus	0	17	0	10	0	0	0	0
Cabestana tabulata	14	3	1	1	3	10	9	3
Crustaceans								
Thais orbita	0	0	0	4	0	0	0	1
Jasus edwardsii	4	9	0	6	3	2	2	4
Plagusia chabrus	0	0	0	0	0	0	0	1

Table X5. Total abundance of fish recorded during volunteer surveys at Crayfish Point in autumn 2001.

Table X6. Total abundance of cryptic fish and invertebrates recorded during volunteer surveys at Tinderbox central (ramp)-site in autumn 1999.

Table X7. Total abundance of fish recorded during volunteer surveys at Tinderbox central (ramp)-site in autumn 1999

Table X8. Total abundance of cryptic fish and invertebrates recorded during volunteer surveys at Piersons Point in autumn 1999.

Table X9. Total abundance of fish recorded during volunteer surveys at Piersons Point in autumn 1999.

Table X10. Total abundance of cryptic fish and invertebrates recorded during volunteer surveys at Tinderbox mid-site in autumn 1999.

Table X11. Total abundance of fish recorded during volunteer surveys at Tinderbox mid-site in autumn 1999.

Table X12. Total abundance of cryptic fish and invertebrates recorded during volunteer surveys at Tinderbox mid-site in autumn 2000.

Table X13. Total abundance of fish recorded during volunteer surveys at Tinderbox mid-site in autumn 2000.

Table X14. Total abundance of cryptic fish and invertebrates recorded during volunteer surveys at Tinderbox mid-site in autumn 2001 (18/2/01).

Table X15. Total abundance of fish recorded during volunteer surveys at Tinderbox mid-site in autumn 2001 (18/2/01).

Table X16. Total abundance of cryptic fish and invertebrates recorded during volunteer surveys at Tinderbox mid-site in autumn 2001 (23/3/01).

Table X17. Total abundance of fish recorded during volunteer surveys at Tinderbox mid-site in autumn 2001 (23/3/01).