Development of a long-term program to monitor coastal communities within the Swan region

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ROTTNEST ISLAND WESTERN AUSTRALIA

Foreword

Progress towards developing explicit metrics for Resource Condition Targets.

This project is one of three that has been undertaken by the Department of Fisheries (DOF), and funded by the Swan Catchment Council (SCC), that aim to gain a better understanding of the biodiversity and community structure within the Swan region. Ultimately, the goal of the SCC projects is to provide information that will allow development of effective and efficient resource condition targets (RCTs).

While fishing is one of the significant factors that needs to be considered when managing coastal marine ecosystems it is not the only driver of change in these communities. Therefore, the various SCC-funded projects undertaken by DOF not only included a focus on targeted species (e.g. Category 1 angling species, blue swimmer crabs, western rock lobster) but they all (including some other non-SCC projects) have focussed on the most appropriate sampling method (in terms of time, accuracy and cost) to generate information on biodiversity so as to provide a measure for general ecosystem health. They have all provided information on the abundance (or relative abundance) and diversity of species from particular categories (e.g. fish, macro-invertebrates), from particular habitats or regions and at particular time intervals (e.g. seasonal comparisons). They have also addressed one of the key issues pertaining to the development of RCTs for biodiversity and community structure, which is to provide baseline information on natural levels of variability.

It has been widely acknowledged that there is a dearth of broad scale ecological studies within the marine ecosystems of WA. This means that these current, or recently completed, studies are essentially establishing baseline descriptions of these communities or assemblages. Consequently, it is not yet possible to set explicit reference points for the management of marine biodiversity because no adequate metrics have been established. This is in contrast to the generally agreed metrics that are now used for the management of individual stocks of exploited fish. For this, the biomass level of a species is often the metric against which the resource condition target (often termed biological reference points, BRPs) is set (e.g. maintain biomass above 40% of the unfished level). The lack of a common metric for measuring biodiversity (or ecosystem health) limits our ability to set meaningful and defensible RCTs. While aspirational RCTs can be developed, to achieve pragmatic management outcomes it is critical that even these are based on a credible scientific understanding or hypothesis if they are to have any real impact on managing marine systems.

In the near future it is likely that the achievable goals for management might include objectives such as to ensure: - no loss of biodiversity; - no change in the community assemblage for a particular group such as fish or algae; - an improvement in habitats or ecosystems deemed to be degraded. Therefore this current suite of studies should be considered as the starting point for the management of biodiversity, not as the end point.

Further work will be required to develop metrics that can "describe" biodiversity and ecosystem structure in a pragmatic and measurable manner. The scope of the current projects did not include the types of comparative tests required to ascertain with confidence which data sets and analyses are most appropriate for developing the required metrics for biodiversity or community structure. Therefore, each of the projects undertaken by DOF for the SCC could undertake further analytical work on the data already available.

Data collected by these three complimentary studies within the Swan region indicates that the habitats within this ecosystem can differ significantly. For example, different categories of benthic cover and demersal scalefish occur at each of the locations examined, which included some areas closed to fishing. Similarly, different beaches along the coast have different assemblages of fish despite the habitats often superficially appearing similar. The ongoing challenge for managing marine ecosystems, therefore, is not only what to measure/monitor, but also at what spatial and temporal scales.

DOF in association with DEC and broader membership of the State Marine Policy Stakeholders Group has been addressing this significant challenge through the development of a risk assessment approach, which is being undertaken within the WAMSI project on ecosystem-based fisheries management (EBFM). This project is identifying all the natural assets within the entire West Coast Bioregion, including the region of specific interest to the SCC. The EBFM project builds on the considerable work undertaken over the past decade to develop a practical system to implement ESD across Australian fisheries. This system, which has full support from all Australian state and federal agencies involved in managing natural marine assets, critically recognises that not all issues (or species, habitats, problems etc.) can be dealt with at a highly detailed level, so the only practical solution is to prioritise issues based on their risks (see www.eafm.com.au for more details).

The risk assessment approach, which forms the basis of the EBFM project, follows nationally agreed standards and methods to help identify priorities. The outcomes from this and the other SCC projects (and other activities focussed on assessing baseline of biodiversity) are now being utilised within the context of the EBFM project, the state's regional marine planning process and any other relevant planning processes. This is being done to ensure the newly acquired information is used to help assess risk status for different habitats within ecosystems as well as to help develop pragmatic metrics for RCTs to underpin the effective management of our marine resources.

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Table of Contents

Figures

Tables

Executive Summary

Marine benthic communities along the metropolitan coast of Western Australia are varied and unique. The southward flowing Leeuwin Current brings warm water and tropical recruits whilst the cooler counter current, the Capes Current, brings the temperate recruits. The result is a mosaic of tropical species intermingled with the dominant temperate species. These areas sustain varied fish and invertebrate communities making this coast a highly desirable location for recreational and commercial use. Management strategies are in place that attempt to maintain the sustainability of this area including various fishery restrictions and the implementation of areas zoned as sanctuary zones (no-take). However, to date, long-term monitoring of the effects of these implementations with respect to the broader ecosystem (not stock specific) is lacking.

The Department of Fisheries (Dof), in working towards an ecosystem based fishery management (EBFM) strategy, is actively seeking to broaden the traditional methods of fishery management, such as species specific bag and size limits, with a more comprehensive assessment of trophic interactions, in order to devise sustainable and environment-friendly fishery methods and goals. To work towards achieving this goal the DoF has undertaken a preliminary marine monitoring program in established marine parks of the Natural Resource Management (NRM) Swan region. This program has gathered baseline data on benthic invertebrate and floral communities, Western Rock Lobster and fish communities (in particular those listed as high risk in west coast fishing rules, i.e. Category 1).

Quantitative baseline data of these fauna and floral groups in areas both fully protected and with limited protection from fishing will allow comparison with future data and identify possible anthropogenic impacts from activities such as fishing. As the majority of the sanctuary zones studied herein were only recently gazetted, only limited positive effects were seen, and only for Western Rock Lobster at one site at Rottnest. Neither the fish nor benthic communities exhibited any differences in terms of numbers of species or their densities as a result of protection from fishing. This highlights the need for long-term monitoring programs of these communities to enable a clearer picture of the real effects of protection and the benefits of the current arrangement of "no fishing" zones in terms of location and size. This information can

then be used by marine planners and mangers to make better informed decisions as to the future use/protection of the marine environment.

Another main objective of this project was to test and develop a robust monitoring regime for the benthic communities, Western Rock Lobster and fish communities, which would allow the identification of indicator species for management and early detection of changes due to anthropogenic pressures.

It is recommended that algal functional groups be used as future indicators of change to the benthic communities, as they require minimal taxonomic expertise (i.e. surrogates can be used). These functional groups are identifiable from close-up (< 1m) digital imagery taken of the substratum, precluding the need to remove vast amounts of material from the marine sites. Digital video footage should be the preferred method of image collection as dive times are less, allowing greater sample replication. Additional collection of digital still images (photoquadrats) every five years will ensure that video footage continues to be the most effective method of capturing information regarding percentage cover of flora and fauna categories. Baseline seasonal information should continue to be collected for the next few years, whereafter (depending on results) sampling may be able to be reduced to annually (e.g. in summer only). The quadrat method should be used to quantify the invertebrate community coupled with testing the effect of lower order taxonomic identification. For all sampling regimes, at each location, replication of transects should be increased.

The Western Rock Lobster project highlighted the importance of a multi-disciplinary approach to determining the relative abundance and size composition of this species. The combination of a potting program, utilising several pot types, with underwater visual census (UVC), provides a more comprehensive assessment of the lobster population, which is essential when using a species such as this as a biological indicator or when assessing the effectiveness of closures to fishing in marine parks. It is recommended that annual surveys be conducted, rather than for example seasonal surveys, as they are more effective and provide a standardised sampling period and consistency between years.

The use of stereo baited remote underwater video (SBRUV) to study fish communities provided unique visual imagery of habitat and fauna, without the need for divers, and important quantitative data, such as relative abundance and a fish length measuring capacity. The response of fish species to the SBRUV and bait affected the likelihood that they were recorded during a particular survey, however, increasing the replication of survey days at sites (i.e. > 2 days within a season) and also increasing spatial replication can overcome this. It was found that camera drops of greater than sixty minutes were needed to survey the relative abundance and diversity of uncommon and unresponsive Category 1 species. This contrasted to only 30 minutes of video footage being required to sample the smaller species. Importantly the smaller inshore sanctuary zones were found to be too small to accommodate the home range of the larger Category 1 species. Further it is recommended that future surveys are undertaken in Spring as this was when greater diversity and abundance of species was recorded.

It is clear from the results of this study that, prior to the creation of a marine park the objectives of the park and sanctuary zones (i.e. what is it aiming to protect) are fundamental considerations. The results from this study highlight the need for future studies to not only monitor changes to the flora and fauna communities in the marine parks of the NRM Swan region, but also to provide empirical evidence as to their effectiveness and as a result allow better informed marine planning decisions

1. General Introduction

1.1. Introduction

Over many decades marine environments have been under pressure from a range of human activities. These pressures include commercial and recreational fishing, coastal development, pollution, introduced pests and the global threat of climate change (Halpern 2003; Pauly et al. 1998). Individually, or in combination, these pressures can contribute to a reduction in viability, significant alteration of an ecosystem, or even habitat loss (Halpern 2003). Species numbers may decline, reducing biodiversity and altering trophic structures (Babcock et al. 1999; Castilla 1999). Pressures such as pollution and climate change present marine managers with difficult challenges as the causes may be diffuse and impacts wide-ranging. Extractive activities such as commercial and recreational fishing have been managed by instigating strategies to deal with changes to their stocks. Currently in Western Australia there are difficulties in estimating participation, total catch or effort by recreational fishers, with no caps on recreational effort. However, strategies such as bag limits, catch limits, minimum legal sizes, licenses, fishing gear restrictions, seasonal closures and fish protection areas go some way to reduce and manage the extractive impacts. These traditional methods of fishery management may be limited in their ability to deliver sustainable outcomes due to limitations in enforcement capabilities or just general lack of knowledge of ecosystem complexity (Castilla 2000). This limitation is becoming more evident with fisheries world-wide facing the prospect of collapse or at least significant depletion of their stocks (Bohnsack 1998; Sumaila et al. 2000), the effects of which have ramifications throughout trophic levels, possibly permanently altering marine ecosystems (see review by Jennings and Kaiser 1998). One way in which managers have sought to mediate these effects is by implementing protective zones i.e. Marine Protected Areas (MPAs) and using them as a complimentary tool to traditional management strategies (Boersma and Parrish 1999; Roberts et al. 2005).

1.2. Marine Protected Areas

The term Marine Protected Area (MPA) is regularly interchanged with marine reserve, marine harvest refugia and marine sanctuary. In this respect the term MPA can be misleading as it may refer to an area where there are few or limited restrictions and hence proffered limited protection, to areas designated as no-take and so fully protected from extractive activities (Boersma and Parrish 1999). However MPA does refer to an area where human activities are managed (i.e. restricted or prohibited) within a spatial area. As such, they provide a physical area in which to enact the precautionary principle, i.e. pre-emptive protection (Agardy 2007; Carr 2000). Thus MPAs provide a spatially explicit approach to management of anthropogenic impacts and as a management tactic can potentially address a broad spectrum of ecological concerns (Carr 2000; Fogarty 1999).

Reasons for creating MPAs include: protecting and maintaining biodiversity; using them to gain scientific understanding of the marine environment; and protection of a functioning community to provide economic, cultural and ethical benefits for future generations (Boersma and Parrish 1999; Castilla 2000). The goal of a MPA, as an ecosystem management strategy, is to protect and maintain marine biodiversity and natural and cultural resources (Allison et al. 2003; Carr 2000;Edgar and Barrett 1999; Sumaila et al. 2000). These protected areas may include seagrass beds, temperate reefs, coral reefs, mangrove systems or other areas that are identified as having significant ecological or cultural value. However, politically driven constraints on design, size and location of MPAs due to conflicting objectives of stakeholders and managers, lack of scientific understanding of species effects and ineffective enforcement of restrictions threaten the effectiveness of these MPAs (Carr 2000; McNeill.S.E 1994).

However, MPAs are as susceptible to the impacts of pollution, coastal development and the spread of introduced species as unprotected areas (Boersma and Parrish 1999; Planes et al. 2000). It is has also been suggested that increased visitation by people to MPAs results in a different suite of effects requiring their own ongoing management strategies (Planes et al. 2000; Rottnest Island Authority 2007). Importantly, as MPAs usually only constitute a fraction of the marine environment they can not be relied upon as the only management tool but rather should be used together with other complimentary management methods (Trexler and Travis 2000). They are, however, the only tool at present which addresses the needs of the whole ecosystem(as long as they are large enough and all negative influences are controlled) rather than being focussed on target species as are traditional fisheries management tools and this is undoubtedly their strength (Jones 2007).

Different zoning schemes are used within MPAs, representing different levels of protection (Department of Environment and Conservation 2007). These different levels of protection allow the separation of conflicting uses whilst still providing areas for commercial, recreational and scientific activities (Department of Environment and Conservation 2007). It is widely acknowledged that there must be some area allocated within an MPA as entirely no-take if the MPA is going to provide refuge for exploited species. These areas are commonly referred to as marine reserves or sanctuary zones (Acosta 2002). Sanctuary zones provide refuge for exploited species and can act as a source of larval and juvenile export to unprotected areas by providing habitat protected from extractive measures (Edgar and Barrett 1999; Francini-Filho and Moura; Manriquez and Castilla 2001). In this sense, sanctuary zones are broad ecosystem-based management tools that provide protection for all marine communities within designated areas (Bohnsack 2000). For a sanctuary zone to meet its management goals the critical issue of size needs to be addressed along with the secondary considerations of efficient enforcement and community acceptance of the protection (Sumaila et al. 2000).

1.3. Sanctuary Zones

1.3.1. Benefits to Biodiversity

First and foremost, sanctuary zones go some way to providing insurance against uncertainty whilst providing a buffer for human impacts (Anon 1999; Boersma and Parrish 1999; Roberts et al. 2005). A review by Halpern (2003) of 89 studies undertaken in marine reserves sanctuary zones were associated with higher values of density, biomass, animal size and diversity compared to fished areas. These expected increases in size in invertebrate populations have been shown in studies by Acosta (2002), Babcock et al. (1999; 2007) and Manriquez and Castilla (2001). Acosta (2002: *Panulirus argus*) and Babcock et al. (1999: *Jasus edwardsii*; 2007: *Panulirus cyngus*) showed this increase in density and size of lobsters in an area as a result of that area being designated a sanctuary zone. Acosta (2002) also found the same for the gastropod queen conch (*Strombus gigas*). Manriquez and Castilla (2001) also found higher densities and greater production of eggs by gastropods in Chilean no-take reserves compared to harvested areas. Various studies on the effects of sanctuary zones on fish populations have given similar results. For example, Friedlander et al. (2003) found higher species richness and diversity in fish assemblages in no-take areas compared to fished areas. Results from a multi-species study into the effects of Tasmanian sanctuary zones by Edgar and Barrett (1999) found fish, invertebrate and algal species numbers were higher, densities of large fishes (>325 mm length) and rock lobsters were greater and the mean size of blue-throated wrasse and abalone were bigger than reference sites in fished areas. These and other numerous studies (see Bohnsack 2000; Goni et al. 2001; Ojeda-Martinez et al. 2007; Russ and Alcala 1996) are starting to scientifically and rigorously quantify the theoretical effects of sanctuary zones and in so doing, making the current and future use of no-take areas less susceptible to criticism and cynicism. Thus the proposed virtues of sanctuary zones are numerous and include:

- Protection of spawning stock (Trexler and Travis 2000)
- Acting as recruitment source for fished areas (Abesamis and Russ 2005; Boersma and Parrish 1999; Bohnsack 1998; Pauly et al. 2002; Roberts et al. 2001; Roberts and Polunin 1991)
- Providing refugia (Boersma and Parrish 1999)
- Maintaining age and size structures of target species (Anon 1999; Bennett and Attwood 1991; Trexler and Travis 2000)
- Protecting genetic resources (Trexler and Travis 2000)
- Potentially providing for a more rapid recovery if there is a fishery collapse (Bohnsack 1998)
- Conservation of non-target species
- Protection of the ecosystem structure, function and integrity (Bohnsack 1998)

1.3.2. Non-fishery benefits

Sanctuary zones also have indirect benefits, such as providing the opportunity for education and research (Bohnsack 1998). Increased scientific knowledge and understanding of the marine ecosystem and the effects of human induced changes provides marine managers with the means to develop well informed strategies for their ongoing management. Further, increasing our understanding of the marine environment can foster the general public's appreciation of the resource and its management, engendering a feeling of responsibility and stewardship. Sanctuary zones also provide an area for non-extractive (non-consumptional) recreational activities thus promoting eco-tourism (viewing of natural ecosystems) with flow on economic benefits (Gell and Roberts 2002; Sobel and Dahlgren 2004; Sumaila et al. 2000). Other economic benefits may be realised by increasing sustainability of resources and hence protecting employment dependent on those resources (Sumaila et al. 2000). Also areas closed to extractive activities can be more easily enforced than other traditional measures and so improve fairness and equity for all (Anon 1999).

1.3.3. Potential costs of sanctuary zones

A known cultural cost of a sanctuary zone is the limitation of certain activities previously allowed, for example fishing. There may also be potential costs (long and short-term) associated with the creation of sanctuary zones. Some of these result from the limited understanding we have of the long-term effects of sanctuary zones, and include the perceptions that they don't work, they add unnecessary regulations, they shift the fishing pressure from closed areas to areas already being exploited and are costly (both in terms of management and fisheries income) (Boersma and Parrish 1999). Another concern is that they will have adverse effects on traditional users by limiting their access to culturally significant fishing stocks (Boersma and Parrish 1999). This needs to be addressed in the early stages of the planning process by ensuring traditional users are included in the decision making process so they aren't made to feel victimised (Gell and Roberts 2002). Most negativity associated with establishment of sanctuary zones (and MPAs) revolve around fishing issues and the perceived reduction in availability of targeted species, however Jones (2007) believes the protection of fishing stocks is a secondary consideration of sanctuary zones and should not cloud the benefits that they have for non-target species. From a management perspective where to site no-take zones and what sizes to make them (so they can achieve their goals) further complicates the decision making process.

1.4. MPA and Sanctuary Zone Design

Prior to the establishment of an MPA (and sanctuary zone) the objectives of the protected area and what it needs (size, restrictions etc.) to be effective must be clearly defined (Acosta 2002; Fogarty 1999; Halpern 2003). It is also necessary that an MPA/sanctuary zone not only protects what is perceived as the critical habitat for marine communities but enough of it, in this sense size is a crucial consideration (Allison et al. 1998). A sanctuary zone that is too small may have limited export function and be more susceptible to catastrophic events (Halpern 2003). If the wrong habitat is protected (i.e. not appropriate for the goals of the MPA) or low quality habitat (i.e. habitat that has diminished ability to provide the desired protection) is chosen then there will be different benefits gained from the closure (Armstrong et al. 1993; Heslinga et al. 1984; Mayfield et al. 2005; Sumaila et al. 2000). These issues of location and size are susceptible to political pressure from different stakeholder groups.

Ultimately the scale of the MPA/sanctuary zone needs to be appropriate for the species, habitat and fisheries they are designed for (Roberts et al. 2005). Examples of considerations that should be addressed include:

- What is to be protected?
- Species dispersal potentials, including larval, juvenile and adult and their habitat requirements at these different life stages (Anon 1999; Boersma and Parrish 1999; Carr 2000; Sumaila et al. 2000)
- Knowledge of home range and migration patterns (Sumaila et al. 2000)
- Effects of under or over grazing should grazer populations change considerably under protection (Planes et al. 2000)
- Where larval settlement is likely to occur (Sumaila et al. 2000)
- Knowledge of the location, distribution and extent of habitats necessary for sustaining the ecosystem (Friedlander et al. 2003)

If any of the above are unknown or if dispersal and migration distances are very high then a network of reserves may be the best management solution. A network of smaller reserves may be preferable to a few large ones, as long as they are sufficiently large enough to retain reproductive populations (Bohnsack 1998). Further, a single reserve is less likely to fulfil all the goals of different stakeholders. In contrast, a network of reserves does inherently have that potential as a result of the cumulative effect of each individual location's strengths (Roberts et al. 2003b). However, whether a single reserve or part of a network of reserves, to be successful in its goal an MPA needs to work in concert with restrictions on extractive activities (Roberts et al. 2003b;Roberts et al. 2003a).

1.5. Monitoring

The only way to reliably measure if a MPA or sanctuary zone are meeting their resource management goals is to use a monitoring program (Bohnsack 1993; Harriott et al. 1999). The results of which allow marine managers to make informed decisions about ongoing management, develop strategies and refine the design and implementation of the reserve as required. For the monitoring program to be scientifically robust an understanding of the functioning ecosystem must be gained within both protected and exploited areas (Babcock et al. 1999).

The design of the monitoring program is vitally important and often one of the most criticised components of science undertaken in MPAs (Castilla 2000). Many authors advocate the use of Before After Control Impact (BACI) designs which allow the quantification of "normal" conditions prior to changes in management (see Claudet et al. 2006; Fraschetti et al. 2002; Lincoln-Smith et al. 2006; Paddack and Estes 2000). However, for a variety of reasons (including lack of funding), often data isn't collected prior to establishment of protection but rather as a consequence of it. Although this lack of knowledge can not be replaced, the collection of long-term data sets, which are needed to separate naturally occurring variation from human induced changes, may mitigate this knowledge gap. Imperative to these studies is the careful selection of study sites (protected and exploited) to ensure similarity in characteristics and the use of multiple control sites (Fraschetti et al. 2002; Guidetti 2002). Most importantly a lack of pre-establishment knowledge should not preclude the future study of possible effects. Further, there is a responsibility of researchers to convey results of their work undertaken in MPAs to policymakers, managers, resource users and interested public in an understandable and practical manner, in order to enhance the success of MPAs and their appropriate ongoing management (Carr 2000).

The very nature of these broad spatially- and temporally-scaled monitoring programs presents its own challenges including securing ongoing funding and support from political groups. Although some positive effects (of protection) may be seen quickly others may not be seen for some time. In their review of 112 studies conducted in marine reserves, Halpern and Warner (2002) found that positive biological responses (e.g. higher densities and mean biomass) were seen in 1 to 3 years of establishment of full protection. On the other hand, McClanahan (2000) suggests that recovery of coral reef ecology after heavy fishing is much slower and somewhere in the order of 5 to 30 years depending on the measure of recovery (McClanahan 2000). Russ and Alcala (2004) found similar time frames (15 to 40 years) were needed for large predatory fish to attain "natural states". Aside from accounting for natural variation there needs to be a clear definition of what constitutes recovery for the target (e.g. fish, coral or ecosystem) before one can decide if a sanctuary zone is effective (McClanahan 2000; Russ et al. 2005). Further, as research starts to consider multi-species effects, results become less clear, making long-term monitoring a necessity before the efficacy of sanctuary zones can be concretely assessed (Boersma and Parrish 1999). Thus, for the effectiveness of MPAs and sanctuary zones to be reliably quantified, long-term monitoring programs need to be established, comparing sites that are protected and exploited (Boersma and Parrish 1999; Carr 2000; Ojeda-Martinez et al. 2007).

1.6. MPAs in Western Australia

Western Australia has approximately 13,500 km of coastline, the State has jurisdiction over approximately 117,887 km^2 of coastal waters with an additional 2,188,647 km^2 in the Exclusive Economic Zone administered by the Commonwealth (Environmental Protection Authority 2008). Approximately 12 % of the State's waters are classified as MPAs, with 2.5 % protected as sanctuary zones. The role of the Marine Parks and Reserves Authority (MPRA), established in August 1997, is to oversee development and management of the marine reserve system and develop policies towards the preservation of these systems (Marine Parks and Reserves Authority 2007). Western Australia's marine reserves form part of the National Representative System of Marine Protected Areas (NRSMPA), part of Australia's obligation as a signatory to the *Convention on Biological Diversity* which came into force on 29th December 1993. In order to achieve their objectives the MPRA uses a program guided by the Comprehensive, Adequate and Representative (CAR) principle, the goal of which is to provide protection to the biodiversity of representative areas of WA's 18 bioregions (Marine Parks and Reserves Authority 2007). A fundamental requirement of this principle is the use of sanctuary (no-take) zones, which the MPRA (2007) states in its Annual Report 1 July 2006 – 30 June 2007 is a major challenge and as yet not adequately addressed along the Western Australian coast when compared against international standards.

Various Acts and Regulations enacted in the States marine parks include:

- The *Wildlife Conservation Act 1950* (WC Act), administered by the Department of Environment and Conservation (DEC), provides legislative protection for flora and fauna across the State's lands and waters
- The *Conservation and Land Management Regulations 2002* provides a mechanism to manage human impacts in marine parks and reserves through enforcement and licensing
- The *Wildlife Conservation Regulations 1970* regulates interaction with fauna and flora through a licensing system
- The *Fish Resources Management Act 1994* (FRM Act) and *Pearling Act 1990* allows the Department of Fisheries WA (DoF) to manage and regulate recreational, and commercial fishing, aquaculture and pearling throughout the State, including in marine parks and reserves
- The *Western Australian Marine Act 1982* and *Navigable Waters Regulations 1983* regulate boating in State waters and apply within marine parks and reserves and is administered by Department for Planning and Infrastructure (DPI) with assistance of DoF and their Fishery and Marine Officers
- The *Environmental Protection Act 1986*, assessed by the Environmental Protection Authority (EPA), regulates any development that may have a significant effect on the environment in or adjacent to marine parks and reserve

On a day-to-day basis the majority of marine reserves located in the NRM Swan region are managed by DEC whilst fishing is regulated by DoF in consultation with DEC. The exception is the Rottnest Island Reserve which has its own management body, the Rottnest Island Authority, however DoF still regulates fishing in the reserve. Other State agencies with statutory responsibilities in the States marine parks include the Western Australian Maritime Museum, Department of Water, Department of Health, Department of Industry and Resources and the Department of Indigenous Affairs (Department of Environment and Conservation 2007).

1.7. Project Aim

Quantifying anthropogenic influences from natural variability in the marine environment requires monitoring areas that are both protected and unprotected from anthropogenic effects. This allows an objective assessment of the local effects of protection and whether the expected social and environmental values (e.g. great diving/fishing opportunities and protection of biodiversity) are being realised, thus assisting in future marine management and conservation. The Natural Resource Management (NRM) Swan region has within its metropolitan waters MPAs that provide the opportunity to scientifically study the impact of extractive activities on target species and associated benthic habitats by developing a robust monitoring program.

There are a number of threats to the status of shallow water marine ecosystems within the NRM Swan region. The most pressing issues in this region are the impacts of human activities (e.g. coastal development and fishing pressure) and the effects of climate change. The increasing impact of both makes it imperative that an effective program is developed to both monitor the structure of biological communities in this region through time, and to determine the source of any changes that may be occurring.

Establishment of a long-term monitoring program involving repeat surveys of the biota within multiple sites through time allows the identification of changes in marine communities. Embracing the ecosystem based fisheries management (EBFM) approach ensures consideration of the broader community and associated habitats, resulting in a more thorough knowledge of the complex interplay between marine species and their environment. Key species found within the shallow water reef communities of the Swan region include recreationally important species such as Western Rock Lobster, WA dhufish and Blue groper. In order to distinguish between potential sources of change to these communities (eg human versus environmental), sites need to be located in areas where impacts from human disturbances are limited, such as 'no-take' sanctuary zones. As such this data will provide an objective assessment of the local effects of marine sanctuary zones and whether the expected social and environmental values are being generated, which will assist in future marine planning processes.

Three Class A Reserves along the Metropolitan coast, within the NRM Swan region, were used to establish a monitoring program for benthic (biodiversity), Western Rock Lobster and fish communities, within sanctuary zones (closed) and outside sanctuary zones (open). Six locations within these three reserves were used for this project, four in the Rottnest Island Reserve and one each in Marmion Marine Park (MMP) and Shoalwater Islands Marine Park (SWIMP). For the purpose of this report the four locations within the Rottnest Island Reserve will be refereed to by their local names i.e. Armstrong Bay (AB: on the northern side), Parker Point (PP: on the southern side), Green Island (GI: on the southern side) and Kingston Reef (KR: on the eastern side) whilst MMP and SWIMP will be retained for their respective locations. Rottnest Islands Marine Reserve locations are in the central offshore area, MMP in the northern coastal area and SWIMP in the southern coastal area. It should be noted that although classified as Class A Reserves and being referred to as Marine Parks certain extractive activities are allowed within the parks (with the exception of sanctuary zones) and are described in detail below. Of further note, all existing and proposed sanctuary zones in these locations are within areas subjected to significant levels of commercial and/or recreational fishing. They are also areas of high social value being utilised by the broader community for other interests such as swimming, snorkelling/diving and reef walking.

The aim of this project was to develop a robust, long-term monitoring program within the NRM Swan region. This was achieved by addressing the following objectives:

1. Provide baseline seasonal quantitative descriptions of shallow-water fish and benthic communities at multiple sites (in marine parks)

1. Develop cost effective, robust monitoring methodologies to be used in a long-term monitoring program that is able to detect changes to these communities through time

2. Provide relevant advice as required to stakeholders

For the purpose of this report "open" sites refer to those sites for which some protection is afforded (see below for further qualification) and "closed" sites refer to sanctuary zones where no extractive activities are allowed.

1.8. Site Description

1.8.1. Coastal Setting

Along the south-west Australian coast swell and wind-induced waves are the major physical force affecting the coastal region. Swell waves originate in the Indian Ocean and arrive from the south-west in summer and west in winter (highest wave energy) (Department of Conservation and Land Management 2002; Hegge et al. 1996; Raffaelli 2000). Inshore wave conditions can be significantly affected by seasonal wind patterns, with storms arriving from the north (turning to westerly) and the sea breeze from the south-west (Lemm and Masselink 1999; Masselink and Pattiaratchi 2001a; Masselink and Pattiaratchi 2001b; Steedman and Craig 1983). The mainly diurnal micro-tide (mean range 0.5 m, max 0.9 m) has minimal impact on the coast (Hodgkin 1958).

Off the south-Western Australian coastline is a series of bathymetrically complex features including reef-island systems and offshore ridges (continuous and discontinuous), whilst in the nearshore there are sand flats, promontories and tombolos (Searle and Semeniuk 1985). An aeolianite limestone reef system, laid down during the Pleistocene and running for 700 km between 33° S and 28° S, causes the oceanic swell to lose energy through a combination of refraction and dissipation and also limits the fetch length for wind wave development (Chape 1984; Pattiaratchi et al. 1995) thus affording the nearshore region some protection.

An unusual oceanographic feature of the western Australian coast is the presence of the Leeuwin Current. This is a low-nutrient, warm-water current that flows down from the topics carrying tropical marine larvae allowing their dispersion further south than would otherwise occur. This warm current is stronger and closer to shore during the cooler months (April – August) and allows water temperatures to remain around 19°C in winter compared to cooler temperatures of ca 15°C closer to the mainland coast (Pearce and Pattiaratchi 1999). This has direct implications for offshore areas such as Rottnest Island as it allows the persistence of tropical species in the temperate waters of south-western Western Australia. In the south, summer winds generate a northward flowing cooler (temperature) current, known as the Capes Current (Pearce and Pattiaratchi 1999). This Capes Current can cause local upwelling of colder and possibly nutrient enriched, waters (Keesing et al. 2006).

1.8.2. Marmion Marine Park

Marmion Marine Park was the States first Marine Park, gazetted in 1987 as a Class A Marine Reserve (Figure 1.1). This park lies within State waters, between Trigg Island and Burns Rock, extending approximately 5.5 km offshore from the high water mark (Department of Conservation and Land Management 2002) (Figure 1.1). There are numerous limestone reefs with complex underwater structures and some small islands located within the marine park. These complex chains of reef act to attenuate the swell wave energy (Department of Conservation and Land Management 2002). Mean sea water temperatures range between 17°C and 21°C for winter and summer, respectively (Department of Conservation and Land Management 2002).

Total areal extent of the park is 9498 ha (Table 1.1). Three zoning frameworks exist in the park: recreational use, general use and sanctuary zones. The General Use Zone makes up 9422 ha (99.2 %) of the park and extractive activities allowed in this zone include:

- commercial and recreational rock lobster fishing,
- commercial and recreational abalone fishing
- recreational rod and line fishing
- spearfishing on snorkel allowed 1.8 km offshore only

The Recreation Zone (Watermans Recreation Area) has an areal extent of 30.9 ha (0.3 % of total park area). The only extractive activity allowed in this zone is recreational rod and line fishing. In 1999, three sanctuary zones were established in which no extractive activities were allowed other than those associated with research projects that require special permits. These sanctuary zones, Boyinaboat Reef (7.4 ha), Little Island (6.1 ha) and The Lumps (27.9 ha), make-up approximately 42 ha, or 0.44 % of the marine park (Department of Conservation and Land Management 2002) (Table 1.1). The Lumps sanctuary zone and adjacent waters were used for this project as this was the largest sanctuary zone in the park (Figure 1.1).

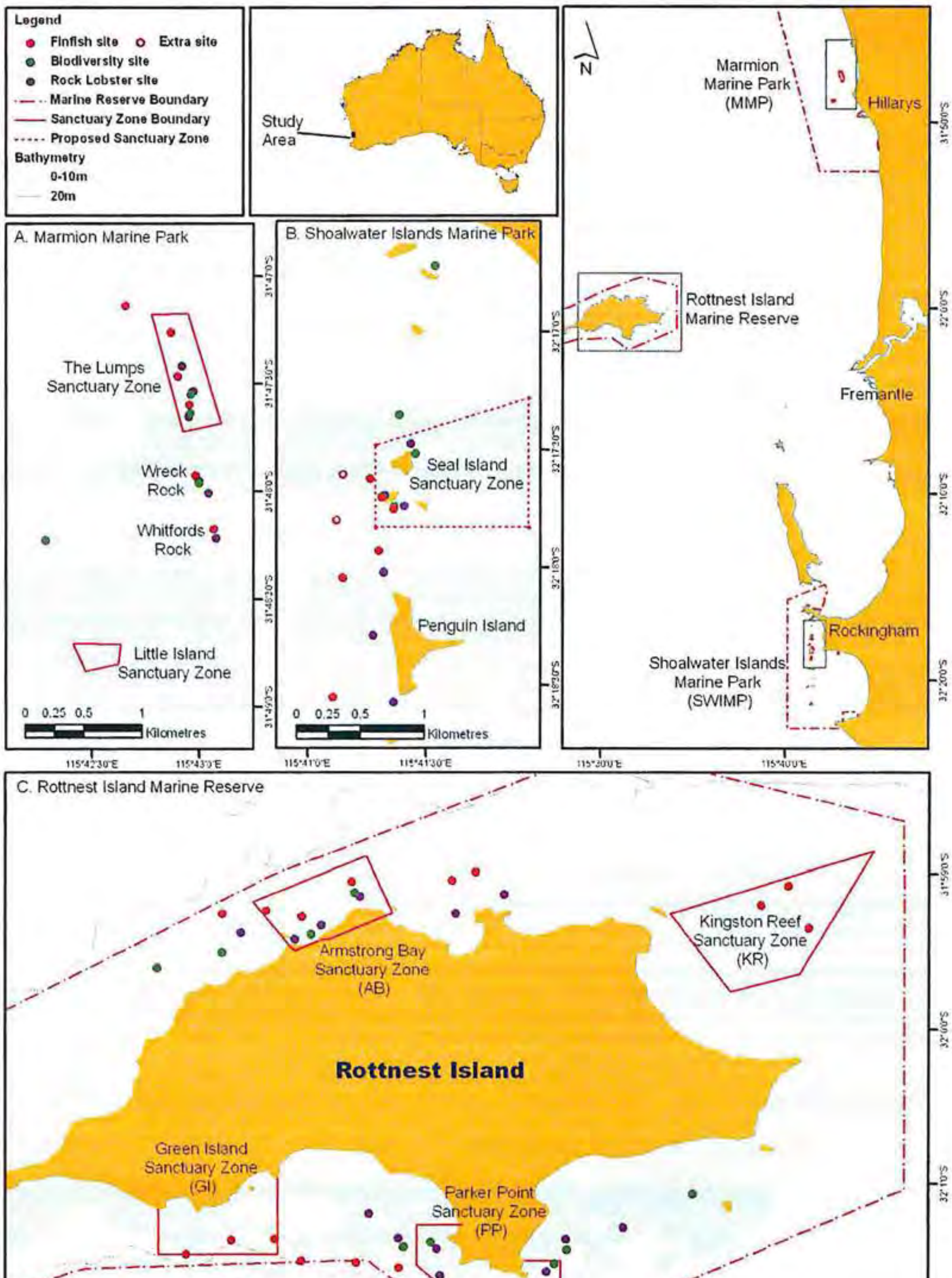

Figure 1.1: The three marine reserves used for developing a long-term monitoring program along the Swan region coast. A. Marmion Marine Park, B. Shoalwater Islands Marine Park and C. Rottnest Island. Sites used for the benthic (biodiversity) studies are indicated by green dots, for rock lobster by purple dots and finfish by red dots.

Table 1.1: Size and year of gazettal of the marine parks and the sanctuary zones sampled during this study.

Marine Parks and their Sanctuary Zones	Size (ha)	Year of Gazettal
Marmion Marine Park	9498	1987
The Lumps SZ	27.9	1999
Shoalwater Islands Marine Park	6658	1990
proposed Seal Island SZ	79*	proposed
Rottnest Island Reserve	3828	1917
Armstrong Bay SZ	82	2007
Green Island SZ	92	2007
Kingston Reef SZ	164	1986, size increased 2007
Parker Point SZ	89	1988, size increased 2007

* This is the proposed size: Department of Environment and Conservation, 2007

The park was identified as a site for protection due to its diverse conservation values that are listed in the Marmion Marine Park Management Plan 1992-2002. Briefly the marine communities and habitats represented within the park were identified as being representative of Western Australia's mid-west coast. Marine habitats include intertidal reef platforms, a high limestone reef 1 km offshore and further offshore (4 km), a limestone reef system (Centaur Reef/Three Mile) known as the Marmion Reefs (Department of Conservation and Land Management 2002).

1.8.3. Shoalwater Islands Marine Park

Shoalwater Islands Marine Park (SWIMP) was gazetted in 1990 as a Class A Marine Reserve (Figure 1.1). This park lies within State waters and is located approximately 50 km south of Perth stretching from Cape Peron in the north to Becher Point in the south (Figure 1.1). The offshore Garden Island Ridge forms an extensive system of intertidal and subtidal limestone reefs and islands in the park. These islands include Penguin Island, Shag Rock, Seal Island, Gull Island, Bird Island, White Rock, The Sisters, Passage Rock, Third Rock, First Rock and Second Rock (Department of Conservation and Land Management 2002). These island and reef systems protect the coastline from offshore swell (up to 90 % attenuated), thus creating relatively low energy coastal lagoons and embayments (Chape 1984; Pattiaratchi et al. 1995). The oligotrophic waters of the park are circulated predominantly by wind-driven means and have an annual temperature range of 15-25 °C (Department of Environment and Conservation 2007; Gordon 1986). The park has two bays, Shoalwater Bay and Safety Bay that are separated by Penguin Island and a deep basin (15 –20 m) in the southern half of the park called Warnbro Sound. Offshore areas of the park are characterised by coarse sediments, while the sediment of the protected basin of Warnbro Sound consists principally of fine carbonate sands and silt, reflecting the relative wave energy (Department of Environment and Conservation 2007).

The marine park has an areal extent of 6658 ha (Department of Environment and Conservation 2007). At present the entire park is categorised as General Use (6658 ha), however a new zoning scheme is proposed for the near future,(Department of Environment and Conservation 2007). The CALM Classified Waters Notice was accepted April 2008 and the Fisheries notice is pending. This new scheme will include general use zones (5681 ha), sanctuary zones (368 ha) and special purpose zones (two areas: Wildlife Conservation and Scientific Reference) (591 ha) (Department of Environment and Conservation 2007). The proposed sanctuary zones will make up approximately 5.5 % of the park's total areal extent. The Shoalwater Islands Marine Park Management Plan 2007 – 2017 proposes the delineation of three sanctuary zones: Seal Island Sanctuary Zone (79 ha); Second Rock Sanctuary Zone (52 ha); and Becher Point Sanctuary Zone (255 ha). Further the Seal Island Sanctuary Zone will be within a Special Purpose Zone (Wildlife Conservation: 425 ha) that aims to protect wildlife (eg sea lions and dolphins) from boat strikes by reducing the boating speed limit to eight knots and prohibiting water skiing (Department of Environment and Conservation 2007).

The Seal Island Sanctuary Zone is populated by little penguin colonies, other breeding and feeding migratory and resident seabirds, Australian seal lion populations and is regularly frequented by bottlenose dolphins (Department of Environment and Conservation 2007). Thus it is an area subject to heavy natural predation of fish communities as well as commercial and recreational fishing pressures. This zone includes areas of seagrass meadows, limestone platforms and reefs dominated by macroalgae. This zone and adjacent waters were used for this project (Figure 1.1).

In the General Use Zone (currently the entire park) the Shoalwater Islands Marine Park Management Plan 2007 – 2017 lists as the extractive activities allowed:

- commercial and recreational rock lobster, crabbing and abalone fishing
- commercial trawling
- commercial and recreational trolling
- commercial netting and line and long-line drop fishing
- recreational set netting, haul and cast/throw netting
- recreational spearfishing on snorkel only
- commercial and recreational collection of aquarium specimens and shells

However, these allowable activities are subject to legislative change. Mineral and petroleum exploration and development is also allowed, at present, within the entire park.

Should the new zoning scheme be formalised and enforced, the restrictions will be as follows.

Within Sanctuary Zones

- no extractive activities, other than those for the purpose of research for which special permits have been issued by DEC

Within the Special Purpose Zone (wildlife conservation) extractive activities allowed include:

- commercial and recreational rock lobster fishing, crabbing and abalone fishing
- commercial aquarium and shell specimen collecting
- commercial vessel fishing
- recreational rod and line fishing and trolling
- recreational octopus potting

Within the Special Purpose Zone (scientific reference) extractive activities allowed include:

- commercial and recreational rock lobster fishing
- recreational rod and line fishing

The extractive activities allowed within the General Use Zone will remain unchanged.

SWIMP was identified as an area for conservation because of its diverse habitats (e.g. seagrass meadows, subtidal and intertidal limestone reef platforms) and mixture of tropical and temperate marine species. Further it provides feeding, breeding and resting grounds for a variety of wildlife including little penguins, Australian sea lions and bottlenose dolphins (Department of Environment and Conservation 2007).

1.8.4. Rottnest Island Reserve

Rottnest Island is a Class A Reserve declared in 1917. The island is located approximately 18 km offshore (west) from Fremantle, Western Australia (Figure 1.1). The marine environment is dominated by patches of high relief subtidal and intertidal limestone reef, seagrass patches and sand. As the island lies in an east-west orientation swell arriving from the south-west has the greatest influence along the south coast and western end especially during south-westerly storms (Wells and Walker 1993). Water temperature varies from 22.5 to 24 $^{\circ}$ C in summer to 18 to 19 $^{\circ}$ C in winter (Pearce et al. 2006).

The reserve waters extend about 800 m from the island, covering 3828 ha in total (Rottnest Island Authority 2007) (Table 1.1). This reserve is also entrusted to the Marine Parks and Reserves Authority (MPRA), however control and management is the responsibility of the Rottnest Island Authority (RIA) established in 1987 under the *Rottnest Island Authority Act 1987* (Rottnest Island Authority 2007). The Department of Fisheries (DoF) manages recreational and commercial fishing within the park, however enforcement is in collaboration with the Rottnest Island Rangers employed by the RIA.

Prior to the 1st July 2007 there were only two sanctuary zones at Rottnest Island: Parker Point (5 ha: established 1988) and Kingston Reef (126 ha: established 1986) (Rottnest Island Authority 2007). Three more sanctuary zones were created and the two existing ones increased in size on 1st July 2007. The new sanctuary zones include Armstrong Bay (82 ha or 2.1 % of total park area), Green Island (92 ha or 2.4 % of total park area) and West End (236 ha of 6.2 % of total park area) with the existing Parker Point and Kingston Reef sanctuary zones being increased to 89 ha (or 2.3 % of total park area) and 164 ha (or 4.3 % of total park area), respectively (Rottnest Island Authority 2007) (Table 1.1). Of the total marine reserve area, sanctuary zones makeup 17.3 %, general use waters 32.3 % and the recreation zone 50.4 %. The sanctuary zones at Armstrong Bay, Green Island, Kingston Reef and Parker Point and adjacent waters were used in this project (Figure 1.1).

Within the sanctuary zones, there are some extractive activities still permitted, these are described individually for each sanctuary:

- Armstrong Bay: rod and/or line fishing allowed from beach at certain sign posted areas.
- Green Island: Shore-based fishing allowed from the Green Island jetty by rod and/or line.
- West End: recreational trolling from boats targeting pelagic species and shore and vessel based recreational line fishing.
- Parker Point and Kingston Reef: are strictly no take.

In the Recreation Zone allowable extractive activities include:

- vessel and shore based line/rod fishing,
- commercial and recreational rock lobster fishing
- commercial netting (except demersal), trawling, beach seine and trap fishing
- commercial collection of aquarium specimens, coral, live rock, shell, sand and mud
- commercial crab and octopus fishing
- spearfishing in most areas unless otherwise sign posted

The marine environment at Rottnest Island is a unique blend of tropical and temperate species, with a large number of endemic species (1993). Assisted by the presence of the Leeuwin Current the island boasts the most southerly occurring tropical coral assemblage in the State (Rottnest Island Authority 2007). For example, the island is the southern-most location of the coral species and genera: *Pocillopora damicornis*, *Porites*, *Alveopora* and *Acropora* (Rottnest Island Authority 2003). Many species-rich seagrass meadows also occur around the island. The reef and seagrass habitats together provide sites for breeding, spawning, feeding and shelter for a vast array of organisms (Rottnest Island Authority 2007).

2. Benthic Communities - Biodiversity

S.D. Bridgwood & T. Coutts

2.1. Introduction

Marine benthic communities along the metropolitan coast of Western Australia are varied and unique. The southward flowing Leeuwin Current brings warm water and tropical recruits whilst the cooler counter current, the Capes Current, brings the temperate recruits (Department of Environment and Conservation 2007). The result is a mosaic of tropical species intermingled with the dominant temperate species. These areas sustain varied fish and invertebrate communities making this coast a highly desirable location for recreational and commercial users. Management strategies aimed at maintaining ecosystem sustainability include various fishery restrictions and the implementation of areas zoned as sanctuary zones (no-take). However, to date, long-term monitoring of the effects of these implementations with respect to benthic communities is lacking.

The aims of this study were firstly to provide baseline descriptions (percentage cover and densities) of benthic communities including algae, seagrass and invertebrates. Secondly to develop a robust monitoring program which would enable the detection of change over time of these communities. To achieve this second aim, different methodologies were trialled. Photoquadrats and video transects were used to determine the percentage cover of different benthic community categories. Quadrat counts and belt transect counts were used to determine the density of invertebrates.

2.2. Methods

The locations used for this study are as described in Chapter 1.8: Site description. Open and closed areas of a location were selected based on visual cues of bottom coverage type, i.e. dominant vegetation, extent of vegetation coverage, reef relief and depth. Within each location (Armstrong Bay: AB, Parker Point: PP, Marmion Marine Park: MMP and Shoalwater Islands Marine Park: SWIMP) 50 m long permanent transects were installed within closed (existing or proposed sanctuary zones) and open sites. Prior to the installation of transects, surveys of potential study sites were undertaken using manta tows and every attempt was made to ensure a locations' open and closed sites were as closely matched as possible. All transects were located in areas that were predominantly limestone reef and oriented in an offshore direction. Each transect was marked at the beginning and end with metal stakes and sub-surface buoys. Six transects were set up for each treatment (i.e. open and closed, n = 12 per location), with the exception of Parker Point which had nine in the open area (i.e. n = 15) (Table 2.1). For the purpose of analyses the open sites at a location were pooled (i.e. averaged across the two sites) as were the closed sites.

Table 2.1: The number of transects located in the open and closed (no fishing) sites for each location. *SWIMP closed site is currently only proposed.

	Open	Closed
Armstrong Bay (AB)	6	6
Parker Point (PP)	9	6
Marmion Marine Park (MMP)	6	6
Shoalwater Islands Marine Park (SWIMP)	6	6*

At Marmion Marine Park (MMP) two closed sites (each with 3 transects) were located within The Lumps Sanctuary Zone (average maximum depth 7.3 m) (see Figure 1.1). The two open sites (each with 3 transects) were located within the marine park in General Use Zones (average maximum depth 6.6 m). Marmion Marine Park (MMP) proved to be the most challenging with respect not only to matching open and closed sites but also within each treatment.

At Shoalwater Islands Marine Park (SWIMP) both closed sites (each with 3 transects) were located within the proposed Seal Island Sanctuary Zone (average maximum depth 5.5 m), however as this is still a proposed zoned (closed site), and to date, the entire park is zoned as general use, these sites were considered open in any analyses, and hence pooled with the other open sites (see Figure 1.1). The proposed Seal Island Sanctuary Zone has within its boundaries a group of five emergent islands (Seal Island and Shag Rock being the largest) that are fringed by intertidal and subtidal limestone reefs (high and low relief), sand and seagrass meadows. Located around the islands is the marine sanctuary zone. The two open sites were located in adjacent waters currently classified as general use (average maximum depth 5.9 m).

Two locations used within the Rottnest Island Reserve were Armstrong Bay (AB) and Parker Point (PP). Armstrong Bay Sanctuary Zone is located on the north side of the island and has intertidal and subtidal limestone reefs (medium to high relief). The two closed sites (each with 3 transects) were located in average maximum depths of 8.5 m (see Figure 1.1). Adjacent control sites (2, each with 3 transects) for AB were located in waters classified as a recreation zone (average maximum depth 7.5 m).

Parker Point Sanctuary Zone is on the southern side of the Rottnest Island and so is more exposed to swell than AB, particularly in winter. Again the area has intertidal and subtidal limestone reefs (average max depth 9.25 m). The closed sites were located in average maximum depths of 8 m (see Figure 1.1). The adjacent control sites for PP were located in waters classified as a recreation zone (average maximum depth 8.7 m).

Fieldwork was undertaken in late winter 2007 and summer 2008. At each location's open and closed sites, 50 m long transects tapes were laid out between the start and end metal stakes. Divers swam along these transect tapes and captured photoquadrats every 10 m from the start to end. Digital video footage was also captured along the full length of the transect. Invertebrates were counted using two different methods, quadrats and belt transects. The methods used to capture photoquadrats, video footage and for counting invertebrates are described in detail below.

2.2.1. Photoquadrat and Video Imagery

Digital still images were captured every 10 m along each transect ($n = 6$) (e.g. Figure 2.1). A stand was used to hold the camera a fixed distance (60 cm) from the substratum for vertical-plane photographs. Use of a wide angle lens (20 mm) on the camera housing resulted in a 50 x 50 cm area of the substratum being captured per image. Multiple images on the same point were taken at each 10 m interval so that the best image (i.e. focussed and free from any 'obstacles' that may hinder analyses) could be used for subsequent analysis. Field time taken to capture the digital photoquadrat images was dependent on the collector but mean dive time ranged between 100 and 180 mins per site.

Video imagery was taken approximately 60 cm above the substratum along the length of the transects, with a 20 mm wide angle lens on the housing, in the vertical-plane. This resulted in approximately the same area of substratum being captured as by the photoquadrat imagery however this was not as consistent as using the frame. Filming rate was as slow and constant as possible with an average of 7 minutes per 50 m transect. Field time taken to capture the video imagery was consistent regardless of the collector, giving dive times of between 30 and 40 mins per site.

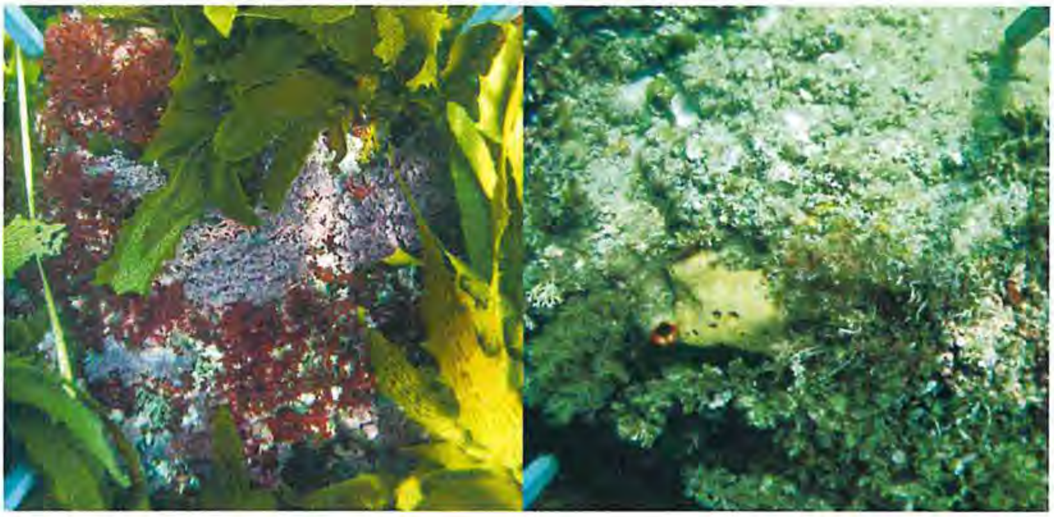

Figure 2.1: Examples of photoquadrat images captured and used for further analyses. The legs of the frame can be seen in the corners of the pictures.

2.2.2. Invertebrates

An ID sheet of photographs of the invertebrates to be counted was developed and used in the field so invertebrates could be counted in-situ (see Appendix 1). As this project was seen as a pilot program where methodologies were tested to determine the most appropriate (in terms of time, accuracy and cost) sampling method to use, two different ways of counting invertebrates were trialled. Both methods were done on SCUBA by the same person to reduce inter-person variability. For both methods broad categories of invertebrates were counted giving a density per metre squared (see Table 2.2). In the first sampling period, invertebrates were counted within a 1 m swathe along the 50 m transect (belt transect) with dive times of up to 120 mins. In the second sampling period, two contiguous 50 x 50 cm quadrats (0.25 m^2) were placed at 5 m intervals along the tape measure, n = 22 quadrats or 5.5 m^2 area. Within each quadrat the substratum was thoroughly inspected for invertebrates. For this method dive times were 60 - 80 minutes.

Table 2.2: The broad category of invertebrates counted for both methodologies trialled, i.e. quadrats and belt transect.

Invertebrate	Classification level
Actiniaria	Order
Ascidiacea	Class
Asteroidea	Class
Scleractinia	Order
Echinoidea	Class
Gastropoda	Class
Holothuroidea	Class
Opisthobranchia	Subclass
Porifera	Phylum
Zoanthidea	Order

2.2.3. Kelp Holdfasts

Following the first sampling period (winter 2007) it was noted that kelp (*Ecklonia radiata*) was the major contributor to algal cover and the study would therefore benefit from further quantifying its contribution to the benthic cover. Thus for the second round of sampling (summer 2008) counting of kelp holdfasts with fronds, was included. Counts were made using two contiguous 0.25 m^2 quadrats at 5 m intervals along each transect giving n = 22 or a 5.5 m^2 area.

2.2.4. Rugosity

A 5 m length of chain was laid down along three sections of each 50 m transect: i.e. at the 0 m, 20 m and 45 m marks. The chain followed the topography of the reef. It was then possible to read along the transect tape as to the distance covered by the chain, thus the chain length (5 m) was divided by the tape distance to give a rugosity value (Pittman et al. 2007). The values for each transect were combined and an average determined for each site then categorised in low, medium or high, i.e. values 1.0 to

1.19 = low, 1.2 –1.49 = medium and > 1.5 = high. Based upon these categories, all locations and their sites were defined as medium to high rugosity (Table 2.3).

Table 2.3: Rugosity of all locations in the closed and open areas. * SWIMP closed site is currently only proposed.

	Open	Closed
AB	high	high
PP	medium	medium
MMP	medium	high
SWIMP	medium	medium*

2.2.5. Data Analyses

2.2.5.1. Photoquadrat and Video Imagery

Photoquadrat and video images were analysed for percentage cover of the following: algae, seagrass, Porifera, Ascidiacea, Bryozoa, Cnidaria, Zoanthidea and Echinoidea. Algae were identified from the images using a combination of their functional morphology and Phylum (i.e. Chlorophyta, Heterokontophyta and Rhodophyta). This resulted in 28 different possible groups. The aim was to assign each specimen of algae to the "best" morphology descriptor possible (see Table 2.4). However if this was not possible (eg poor quality images) they were assigned to a higher order functional group, i.e. frondose, foliose, encrusting, leathery or turfing.

Captured photoquadrat and video imagery was imported into TransectMeasure Version 1.24 (SeaGIS Pty Ltd). This software aids in the analyses of video and photoquadrat images. A designated number of points can be laid over each image and each point can then have various attributes assigned to them. Pilot studies were initially undertaken to determine the optimum number of points required per image (photoquadrat and video) to maximise detail without over processing. The optimum number of points for both was 12, arranged in a 4 x 3 grid. For the photoquadrat images this resulted in 432 points per site being analysed, the exception being PP open which was 648 points due to the additional open site. A pilot study was also run on the video imagery to determine how many frames needed to be captured again for maximum detail but avoiding over processing. The optimum was 41 frames per video length, with a minimum of 150 frames between each sampled frame to prevent overlap. This resulted in 2952 points per site being analysed, again the exception being PP (4428 points). Each point had the following information recorded: date filmed; location; transect number; fishing status (i.e. open or closed); name of the person doing the analysis; latitude and longitude; rugosity; taxa; and functional morphology.

Prior to analysing any imagery all processors were tested against a benchmark series of images to ensure there was good agreement (> 80 %). This level of agreement was maintained throughout the processing. Time taken to analyse one image varied according to how complex the photograph was, taking anywhere between 5 and 15 minutes.

The sampling design consisted of three factors: season (2 levels, fixed: summer and winter), location (4 levels, fixed: Marmion Marine Park, Shoalwater Islands Marine Park, Armstrong Bay and Parker Point) and fishing status (2 levels, random, nested within location and season: closed and open).

Differences in taxa assemblages between season, fishing status and locations were examined graphically with non-metric multidimensional scaling (nMDS), using the Bray-Curtis similarity matrix constructed on square root transformed data (Clarke 1993). Permutational analysis of multivariate dispersions (PERMDISP) was used to test the multivariate dispersions among the groups and is only reported if any failed (Anderson 2004). This method is analogous to a Levene's test, in that it tests the homogeneity of multivariate dispersions. All multivariate analyses were conducted using permutational multivariate analysis of variance (PERMANOVA) in the PRIMER v6 with PERMANOVA statistical package (Anderson 2001; Clarke et al. 2006). Initially, PERMANOVA (9999 permutations) was conducted on the complete sampling design set using the square root transformed Bray-Curtis similarities matrix. If a significant difference in location was found, pairwise comparisons were made during the *a posteriori* analyses using PERMANOVA (9999 permutations) to determine where differences were occurring. Analysis of Similarity (ANOSIM) for significant results between locations was run as a comparison to the *a posteriori* PERMANOVA routine.

Similarity Percentages (SIMPER) analysis (square root transformed data using Bray-Curtis similarity matrix) was used to determine which taxa contributed to seasonal variations (Clarke 1993). Taxa discriminating between the two seasons were analysed separately using one-way ANOVA. Assumptions of ANOVA were checked prior to analysis using Levene's Test. Data not meeting assumptions were transformed. If transformation was not adequate then the nonparametric Mann-Whitney U test was run. Seasons were then separated and SIMPER analysis was used to examine which taxa contributed to location variations. Again differences in percentage cover of categories discriminating between the four locations were analysed separately using one-way ANOVA. As with previous analyses assumptions of ANOVA were checked prior to analysis using Levene's Test, transformations were undertaken as required and if normality wasn't achieved nonparametric Mann-Whitney U test was used.

A canonical analysis of principal coordinates (CAP) (Anderson and Willis 2003) was also performed separately for significant factors with greater than 2 levels (i.e. location), to show whether different taxa could be discriminated between locations. CAP analyses were conducted using the original data. The significant one-way ANOVA results identified from SIMPER were plotted on the CAP analysis using the Pearson correlations of square root transformed data with canonical axes.

Table 2.4: Functional morphology groups used to categorise algae in the photoquadrat and video imagery.

Functional Morphology	Description	Examples (genera)	References
Articulated calcareous (ART)	Calcareous algae with obvious joints, erect with articulated thallus	*Halimeda, Haliptilon, Jania, Amphiroa, Metamastophora*	Steneck, RS & Dethier, MN (1994) Mclanahan, TR et al (2003) Phillips et al. (1997)
Corticated foliose (COF)	Flattened appearance	*Dictyomenia, Plocanium, Gellidium, Pterocladia*	Phillips et al. (1997)
Corticated terete (COT)	Circular in cross section	*Codium, Champia*	Phillips et al. (1997)
Encrusting (ENC)	Stony, calcified, pink crusts, algae form flat expanses over rocks, or on other plants and mollusc shells	*Lithophyllum, Hydrolithon*	Steneck, RS & Dethier, MN (1994)
Filamentous (FIL)	Filamentous algae is described as a row of cells, placed end to end to produce a hair like frond.	*Cladophora, Hincksia*	Steneck, RS & Dethier, MN (1994)
Foliose (FOL)	Foliose algae are distinguished from other forms by the leafy parenchymous sheets of cells.	*Ulva, Kallymenia, Dictyota*	Steneck, RS & Dethier, MN (1994)
Frondose (FRO)	Algae with erect, gelatinous, stiff or bushy thallus	*Padina, Sargassum (recruit), Asparagopsis, Hypnea, Dictyopteris*	Mclanahan, TR et al. (2003)
Hollow/Tubular (HOL)	Hollow or tubular algae lacking distinct internal structure	*Gloiosaccion, Colpomenia, Botryocladia*	Phillips et al. (1997)
Leathery (LEA)	Leathery macrophytes	*Ecklonia, Sargassum, Turbinaria, Betaphycus*	Steneck, RS & Dethier, MN (1994) Phillips et al. (1997)
Turfing (TUR)	Turf algae are a multispecific assemblage of diminutive, often filamentous, algae that attain a canopy height of only 1 to 10 mm		Mclanahan,TR et al. (2003)

2.2.5.2 Invertebrates

For both the quadrat and belt transect methodologies differences in taxa assemblages between fishing status (open and closed) and locations were examined graphically with non-metric multidimensional scaling (nMDS), using the Bray-Curtis similarities on square root transformed data (Clarke 1993). The sampling design included two factors: location (4 levels, fixed: Marmion Marine Park, Shoalwater Islands Marine Park, Armstrong Bay and Parker Point) and fishing status (2 levels, random, nested within location: closed and open). PERMDISP was used to test the multivariate dispersions among the groups and is only reported if any failed (Anderson 2004). For both methodologies, a PERMANOVA (9999 permutations) was run using the square root transformed Bray-Curtis similarities matrix. If a significant difference in location was found, pairwise comparisons were made during the *a posteriori* analyses using PERMANOVA (9999 permutations) to determine where differences were occurring. ANOSIM for significant results between locations was run as a comparison to the *a posteriori* PERMANOVA routine.

SIMPER analysis was used to examine which taxa contributed to location variations. Again differences in biomass of taxa discriminating between the four locations were analysed separately using one-way ANOVA. As before, assumptions of ANOVA were checked prior to analysis using Levene's Test, transformations were undertaken as required and if normality wasn't achieved the nonparametric Mann-Whitney U test was used.

A canonical analysis of principal coordinates (CAP) (Anderson and Willis 2003) was also performed separately for significant factors with greater than 2 levels i.e. to show whether different taxa could be discriminated between locations. CAP analyses were conducted using the original data. The significant one-way ANOVA results identified from SIMPER were plotted on the CAP analysis using the Pearson correlations of square root transformed data with canonical axes.

2.2.5.3 Kelp Holdfasts

The sampling design included two factors: location (4 levels, fixed: Marmion Marine Park, Shoalwater Islands Marine Park, Armstrong Bay and Parker Point) and fishing status (2 levels, random, nested within location: closed and open). PERMDISP was used to test the multivariate dispersions among the groups and is only reported if any failed (Anderson 2004). For both methodologies a PERMANOVA (9999 permutations) was run on an untransformed Bray-Curtis similarities matrix. If a significant difference in location was found, pairwise comparisons were made during the *a posteriori* analyses using PERMANOVA (9999 permutations) to determine where differences were occurring.

2.3. Results

2.3.1. Captured Images

2.3.1.1. Photoquadrats

Percentage cover of the flora and fauna categories from the photoquadrat images were significantly different for season and location (PERMANOVA, $P < 0.05$: Table 2.5) but not for fishing status (i.e. closed, open), nor was there any interaction between season and location (Table 2.5). For season SIMPER revealed that this difference could be attributed to seven flora and fauna categories, all algal functional groups, two of which were significantly different, leathery brown algae and frondose algae, both of which were dominant in summer (1-way ANOVAs and Mann-Whitney U Tests Table 2.6). Pair-wise comparisons showed in summer that there were more leathery brown algae, frondose red algae and frondose algae, whilst frondose brown and green algae, articulated red algae and turfing algae were dominant in winter (Table 2.6).

There was significant variation among locations with all sites being different to each other except MMP and SWIMP (*a posteriori* PERMANOVA Table 2.7). A comparative test using ANOSIM also showed this variation among locations (Global R 0.374, $P < 0.0001$) with the addition of a difference between MMP and SWIMP (Table 2.8). This difference between locations could be attributed to 12 flora and fauna categories in summer, of which 13 pair-wise comparisons were significant (SIMPER, 1-way ANOVAs and Mann-Whitney U Tests Table 2.9). In winter, the location differences could be attributed to 18 categories of which 20 pair-wise comparisons were significant (1-way ANOVAs and Mann-Whitney U Tests Table 2.10). These significant differences were plotted on the CAP graphs so any drivers of patterns could be visualised.

The degree of uniqueness in flora and faunal composition for the locations in summer is illustrated by the leave-one-out allocation success rate from the CAP analysis that was >71 %. A clear separation of the locations driven by category cover types is shown in Figure 2.2 a, with AB being separated from the other locations as it had more brown leathery algae and less of the other categories, SWIMP was separated from the other locations due to a greater presence of seagrass and red frondose algae, MMP by the presence of solitary Ascidiacea and green foliose algae and PP by the red articulating and corticated foliose algal types (Figure 2.2 a).

The uniqueness in flora and faunal composition for the locations in winter was poor compared to summer resulting in a leave-one-out allocation success rate from the CAP analysis of only >47 %. There was more overlap between the Rottnest sites AB and PP than seen in summer, however these two sites were clearly separated from the inshore sites of MMP and SWIMP. The red articulating algae tended to separate the Rottnest sites (AB and PP) from MMP and SWIMP. Solitary Ascidiacea as well as colonial Zoanthidea were useful to separate MMP, and seagrass and red frondose algae to separate SWIMP, from the other locations (Figure 2.2 b).

Table 2.5: PERMANOVA of the differences in the categories for season, location and fishing status from both methodologies, i.e. photoquadrats and video.

	Source	*df*	MS	F	*P* (perm)
Photoquadrats	Season	1	1650.6	2.3306	0.0127*
	Location	3	1783.2	2.5088	0.0006***
	Season x location	3	431.21	0.7234	0.8664
	Fishing (season x location)	6	620.59	1.172	0.2133
	Error	20	529.52		
Video	Season	1	1493.3	4.8662	0.0008***
	Location	3	1630.6	5.3048	0.0001***
	Season x location	3	311.0	1.0940	0.3879
	Fishing (season x location)	7	282.3	0.9779	0.5155
	Error	19	288.63		

Data were square root transformed and analysis based on the Bray-Curtis dissimilarity measure, permutated 9999 times. Significance level is indicated by $*P < 0.05$, $**P < 0.01$ and $***P < 0.001$, *df* degrees of freedom and *MS* mean squares

Table 2.6: Categories identified by SIMPER as typifying the seasonal composition of the photoquadrat images in summer and winter (shaded boxes) and those that distinguished between the seasons (non-shaded boxes). For the pair-wise comparisons the superscript indicates in which season, summer (s) or winter (w), percentage cover was highest.

	Summer	Winter
Summer	Algae Brown leathery Algae Red articulated Algae Red corticated foliose Algae Red frondose Sediment Turfing algae	
Winter	Algae Brown leathery ^{S***} Algae Brown frondose ^{W} Algae Green foliose ^{W} Algae Red articulated ^{W} Algae Red frondose ^{S} Algae frondose $^{S+}$ Turfing algae ^{W}	Algae Brown leathery Algae Red articulated Algae Red corticated foliose Algae Red encrusting Algae Red frondose Turfing algae Sediment

Level of ANOVA significance indicated by $*P < 0.05$, $**P < 0.01$ and $***P < 0.001$ and Mann-Whitney U test $^{+}P < 0.05$.

Table 2.7: An *a posteriori* pair-wise PERMANOVA comparing flora and fauna categories among locations for the photoquadrats and video imagery.

	Groups	*t*	*P* (perm)	*P* (Monte-Carlo)	No. unique values
Photoquadrats	A, M	1.5876		0.0345*	315
	A, P	1.5400	0.0385*		9930
	A, S	2.3536	0.0006***		9932
	M, P	1.5290	0.0328*		9946
	M, S	1.2153	0.2197		9959
	P, S	1.6356	0.0309*		9940
Video	A, M	1.8906		0.0230*	315
	A, P	2.1735	0.0021**		9943
	A, S	2.5680	0.0017**		9942
	M, P	2.5387	0.0026**		9951
	M, S	2.0024	0.0096**		9954
	P, S	2.1778	0.0021**		9939

Number of permutations = 9999, the Monte-Carlo *P*-value was used when there were small number of unique values. Data were square root transformed and based on the Bray-Curtis dissimilarity measure. Level of significance indicated by $*P < 0.05$, $**P < 0.01$ and $***P < 0.001$. *A* Armstrong Bay, *M* Marmion Marine Park, *P* Parker Point, *S* Shoalwater Islands Marine Park

Table 2.8: ANOSIM pair-wise comparisons of flora and fauna categories among locations for the photoquadrats and video imagery.

	Groups	*R*	*P* (perm)	Actual Permutations	Number >= Observed
Photoquadrats	A, M	0.292	0.0040**	6435	28
	A, P	0.202	0.0310*	9999	312
	A, S	0.625	0.0002***	6435	1
	M, P	0.382	0.0003***	9999	2
	M, S	0.307	0.0040**	6435	23
	P, S	0.455	0.0004***	9999	5
Video	A, M	0.320	0.0100**	6435	67
	A, P	0.214	0.0180*	9999	174
	A, S	0.784	0.0002***	6435	1
	M, P	0.337	0.0090**	9999	92
	M, S	0.554	0.0005***	6435	3
	P, S	0.499	0.0003***	9999	2

Number of permutations = 9999. Data were square root transformed and based on the Bray-Curtis dissimilarity measure. Level of significance indicated by $*P < 0.05$, $**P < 0.01$ and $***P < 0.001$. *A* Armstrong Bay, *M* Marmion Marine Park, *P* Parker Point, *S* Shoalwater Islands Marine Park. *R* R statistic

Table 2.9: Categories identified by SIMPER as characterising the flora and fauna composition within locations for the photoquadrat images in summer (shaded boxes) and those that distinguished between the locations (non-shaded boxes). For the pair-wise comparisons the superscript indicates in which location, Armstrong Bay (A), Parker Point (P), Marmion Marine Park (M) and Shoalwater Islands Marine Park (S), the percentage cover was highest.

	Armstrong Bay	Parker Point	Marmion Marine Park	Shoalwater Islands Marine Park
Armstrong Bay	Algae Brown leathery Algae Red articulated Algae Red corticated foliose Algae Red frondose Turfing Algae Sediment			
Parker Point	Algae Brown leathery ^{A}** Algae Brown Filamentous ^{A} Algae Red articulated ^{P}** Algae Red encrusting ^{P} Algae Red frondose ^{P}** Sediment ^{A}	Algae Brown leathery Algae Red articulated Algae Red corticated foliose Algae Red frondose Sediment		
Marmion Marine Park	Algae Brown leathery ^{A} Algae Brown Filamentous ^{A} Algae Green foliose ^{M} Algae Red encrusting ^{M} Algae Red frondose ^{M}** Ascidian solitary ^{M}*** Turfing Algae ^{A}	Algae Green foliose ^{M}* Algae Red corticated foliose ^{P}++ Ascidian solitary ^{M}***	Algae Brown leathery Algae Green foliose Algae Red articulated Algae Red corticated foliose Algae Red encrusting Algae Red frondose Turfing Algae	
Shoalwater Islands Marine Park	Algae Brown leathery ^{A} Algae Brown Filamentous Algae Brown Frondose ^{A} Algae Red encrusting ^{S} Turfing Algae ^{A} Seagrass ^{S}*	Algae Red articulated ^{P}** Algae Red encrusting ^{P} Seagrass ^{S} Sediment ^{S}* Turfing Algae ^{P}	Algae Brown frondose ^{S} Algae Brown leathery ^{S} Algae Green foliose ^{M} Algae Red articulated ^{M}+ Turfing Algae ^{M} Ascidian solitary ^{M}***	Algae Brown leathery Algae Red corticated foliose Algae Red frondose Sediment

Level of ANOVA significance indicated by $*P < 0.05$, $**P < 0.01$ and $***P < 0.001$ and Mann-Whitney U test $^{+}P < 0.05$

Table 2.10: Categories identified by SIMPER as characterising the flora and fauna composition within locations for the photoquadrat images in winter (shaded boxes) and those that distinguished between the locations (non-shaded boxes). For the pair-wise comparisons the superscript indicates in which location, Armstrong Bay (A), Parker Point (P), Marmion Marine Park (M) and Shoalwater Islands Marine Park (S), the percentage cover was highest.

	Armstrong Bay	Parker Point	Marmion Marine Park	Shoalwater Islands Marine Park
Armstrong Bay	Algae Brown leathery Algae Red articulated Algae Red encrusting Algae Red corticated foliose Turfing Algae Sediment			
Parker Point	Algae Brown leathery ^{A} Algae Red corticated terete ^{P} Algae Red encrusting ^{P} Turfing Algae ^{A*} Sediment ^{P} Porifera ^{A}	Algae Brown leathery Algae Red articulated Algae Red corticated foliose Algae Red encrusting Algae Red frondose Turfing Algae Sediment		
Marmion Marine Park	Algae Brown foliose ^{A} Algae Green frondose ^{A***} Algae Red encrusting ^{M***} Algae Red articulated ^{A} Algae Red corticated foliose ^{A} Algae Red foliose ^{A***} Algae Red leathery ^{A} Ascidian solitary ^{M} Zoanthidea ^{M***}	Algae Brown leathery ^{M} Algae Green corticated terete ^{P} Algae Green frondose ^{M} Algae Red articulated ^{P*} Algae Red foliose ^{M***} Turfing Algae ^{M} Ascidian solitary ^{M***} Zoanthidea ^{M***}	Algae Brown leathery Algae Red corticated foliose Algae Red encrusting Algae Red frondose Turfing Algae Sediment Porifera	
Shoalwater Islands Marine Park	Algae Brown foliose ^{A***} Algae Green foliose ^{S} Algae Green frondose ^{S***} Algae Red articulated ^{A} Algae Red corticated foliose ^{A} Algae Red corticated terete ^{S} Algae Red frondose ^{S} Turfing Algae ^{A*} Seagrass ^{S**} Sediment ^{S}	Algae Brown leathery ^{S} Algae Green corticated terete ^{P***} Algae Green foliose $^{S++}$ Algae Red articulated ^{P**} Algae Red encrusting ^{M} Algae Red frondose ^{S**} Turfing algae ^{P} Seagrass ^{S}	Algae Brown frondose ^{M} Algae Green foliose $^{S++}$ Algae Red encrusting ^{M} Algae Red frondose ^{S} Turfing algae ^{M} Ascidian solitary ^{M***} Seagrass ^{S} Sediment ^{S**} Porifera ^{M} Zoanthidea ^{M}	Algae Brown leathery Algae Red frondose Algae Red corticated foliose Algae Red encrusting Sediment

Level of ANOVA significance indicated by $*P < 0.05$, $**P < 0.01$ and $***P < 0.001$ and Mann-Whitney U test $^{+}P < 0.05$

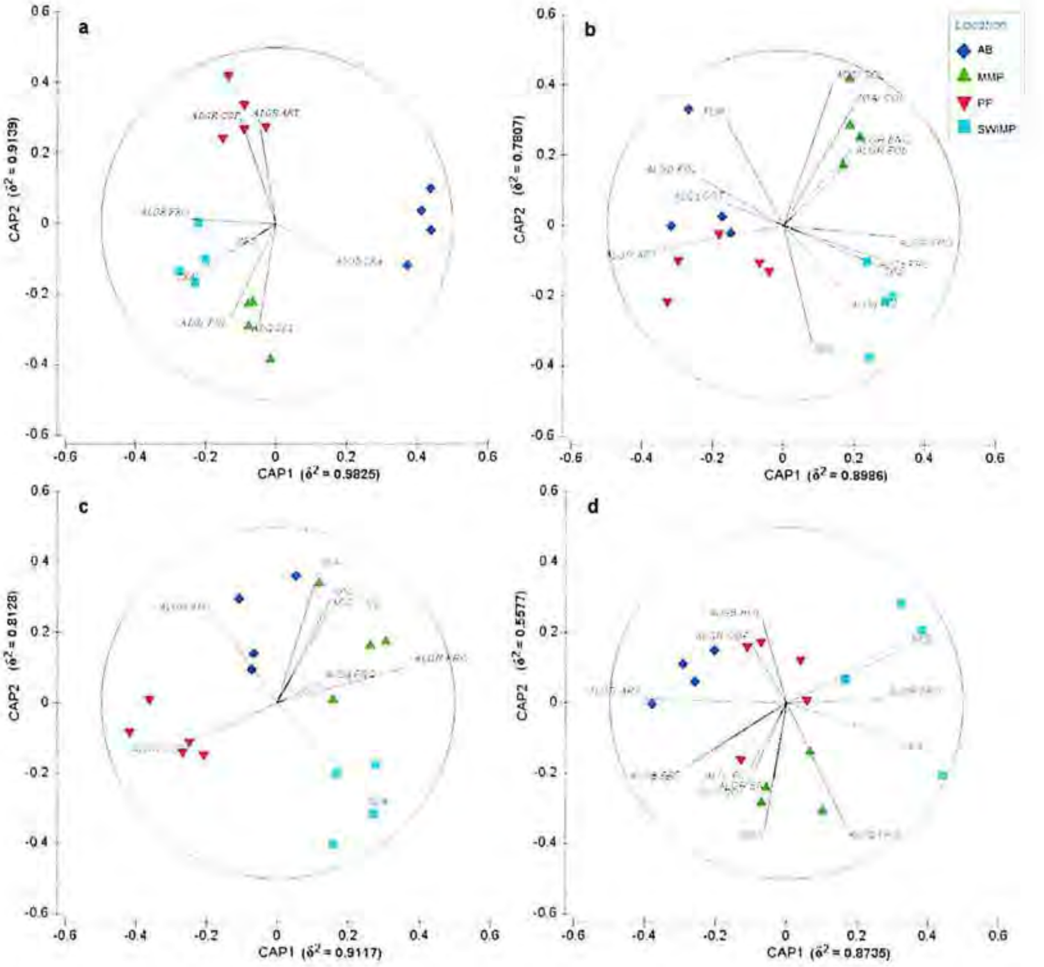

Figure 2.2: Canonical analysis of principal coordinates (CAP) ordination for locations and biplots for a) photoquadrat analysis in summer b) photoquadrat analysis in winter c) video analysis in summer and d) video analysis in winter. Categories include: ASC/ SOL: solitary Ascidiacea, SEA: seagrass, SED: sediment, SPO: Porifera, ZOA/ COL: colonial Zoanthidea, and for algae: ALG/ FIL: filamentous, ALGB FRO: brown frondose, ALGB LEA: brown leathery, ALGB HOL: brown hollow, ALGg FRO: green frondose, ALGg COT: green corticated terete, ALGR ART: red articulated, ALGR COF: red corticated foliose, ALGR ENC: red encrusting, ALGR FOL: red foliose, ALGR FRO: red frondose, TUR: turfing.

2.3.1.2. Video

As with the photoquadrat imagery the percentage cover of the different flora and fauna categories from the video analysis were significantly different for season and location (PERMANOVA, $P < 0.05$: Table 2.5). There was no significant difference for fishing status (i.e. open or closed) or for the interaction (season x location). For season, SIMPER revealed that this difference could be attributed to seven flora and fauna categories, including six algal functional groups, three of which were significantly different: brown leathery algae (dominant in summer), red encrusting algae and turfing algae (both dominant in winter) (1-way ANOVAs and Mann-Whitney U Tests Table 2.11). For the pair-wise comparison, leathery brown algae and frondose algae were dominant in summer, whilst articulated, encrusting and corticated foliose red algae, turfing algae and sediment were more prevalent in winter (Table 2.11).

There was significant variation among locations with all sites being different to each other (*a posteriori* PERMANOVA Table 2.7). A comparative test using ANOSIM also showed this significant variation among locations (Global R 0.447, P < 0.0001, Table 2.8). SIMPER revealed differences between locations could be attributed to 16 flora and fauna categories in summer, of which, 21 pair-wise comparisons were significant (1-way ANOVAs and Mann-Whitney U Tests Table 2.12). In winter the location differences could be attributed to 18 categories, of which 29 pair-wise comparisons were significant (1-way ANOVAs and Mann-Whitney U Tests Table 2.13). These significant differences were plotted on the CAP graphs so any drivers of patterns could be visualised.

Table 2.11: Categories identified by SIMPER as typifying the seasonal composition of the video images in summer and winter (shaded boxes) and those that distinguished between the seasons (non-shaded boxes). For the pair-wise comparisons the superscript indicates in which season, summer (s) or winter (w), the percentage cover was highest.

	Summer	Winter
Summer	Algae Brown leathery Algae Brown frondose Algae Red articulated Algae Red frondose Turfing Algae Sediment	
Winter	Algae Brown leathery ^{S***} Algae Red articulated ^{W} Algae Red corticated foliose ^{W} Algae Red encrusting ^{W***} Algae frondose ^{S} Turfing Algae ^{W**} Sediment ^{W}	Algae Brown leathery Algae Brown frondose Algae Red articulated Algae Red frondose Algae Red encrusting Turfing Algae Sediment

Level of ANOVA significance indicated by $*P < 0.05$, $**P < 0.01$ and $***P < 0.001$

Table 2.12: Categories identified by SIMPER as characterising the flora and fauna composition within locations for the video images in summer (shaded boxes) and those that distinguished between the locations (non-shaded boxes). The superscript indicates in which location, Armstrong Bay (A), Parker Point (P), Marmion Marine Park (M) and Shoalwater Islands Marine Park (S), percentage cover was highest.

	Armstrong Bay	Parker Point	Marmion Marine Park	Shoalwater Islands Marine Park
Armstrong Bay	Algae Brown leathery Algae Red articulated Algae Red frondose Algae frondose Turfing Algae Sediment			
Parker Point	Algae Red corticated foliose ^{P} *** Algae Red encrusting ^{P} Algae Red frondose ^{A} *** Algae filamentous ^{A} Turfing Algae ^{A} Sediment ^{A} Porifera ^{A}	Algae Brown leathery Algae Red corticated foliose Algae Red articulated Turfing Algae Sediment		
Marmion Marine Park	Algae Brown leathery ^{A} Algae Red articulated ^{A} ** Algae Green frondose ^{M} * Algae Green foliose Algae filamentous Algae frondose ^{M} Turfing Algae ^{M} ** Ascidian solitary ^{M} Sediment ^{A} Porifera ^{M}	Algae Brown frondose ^{M} Algae Brown leathery ^{P} + Algae Red frondose ^{M} *** Algae Green foliose ^{M} Algae Green frondose ^{M} Algae filamentous ^{M} Turfing Algae ^{M} *** Ascidian solitary ^{M} *** Sediment ^{M} Porifera ^{M}	Algae Brown frondose Algae Brown leathery Algae Green frondose Algae Red articulated Algae Red encrusting Algae Red frondose Turfing Algae Sediment	
Shoalwater Islands Marine Park	Algae Green foliose ^{S} Algae Red articulated ^{A} ** Algae Red corticated foliose ^{A} * Algae Red leathery ^{A} Algae filamentous ^{A} Algae frondose ^{A} Turfing Algae ^{A} * Sediment ^{S} Seagrass ^{S} ** Porifera ^{A} ***	Algae Brown frondose ^{S} Algae Green foliose ^{S} Algae Red articulated ^{P} * Algae Red corticated foliose ^{P} Algae Red frondose ^{S} *** Turfing Algae ^{P} Seagrass ^{S} Sediment ^{S}	Algae Brown frondose ^{M} Algae Brown leathery ^{S} Algae Green frondose ^{M} Algae Red articulated ^{M} ** Algae Red corticated foliose ^{M} * Algae filamentous ^{M} Turfing Algae ^{M} *** Ascidian solitary ^{M} *** Sediment ^{S} Seagrass ^{S} Porifera ^{M} ***	Algae Brown frondose Algae Brown leathery Algae Red frondose Sediment

Level of ANOVA significance indicated by $*P < 0.05$, $**P < 0.01$ and $***P < 0.001$ and Mann-Whitney U test $^{+}P < 0.05$

Table 2.13: Categories identified by SIMPER as characterising the flora and fauna composition within locations for the video images in winter (shaded boxes) and those that distinguished between the locations (non-shaded boxes). The superscript indicates in which location, Armstrong Bay (A), Parker Point (P), Marmion Marine Park (M) and Shoalwater Islands Marine Park (S), percentage cover was highest..

	Armstrong Bay	Parker Point	Marmion Marine Park	Shoalwater Islands Marine Park
Armstrong Bay	Algae Brown frondose Algae Brown leathery Algae Red articulated Algae Red encrusting Algae Red frondose Turfing Algae Sediment			
Parker Point	Algae Brown frondose ^{A} Algae Brown hollow ^{A} Algae Red articulated $^{A *}$ Algae Red corticated foliose $^{P **}$ Algae Red encrusting ^{P} Algae Red frondose $^{P *}$ Algae filamentous $^{A *}$ Turfing Algae $^{A ***}$ Sediment $^{P *}$	Algae Brown leathery Algae Red corticated foliose Algae Red encrusting Algae Red frondose Turfing Algae Sediment		
Marmion Marine Park	Algae Brown hollow $^{A ***}$ Algae Brown leathery ^{A} Algae Green frondose $^{M ***}$ Algae Red articulated $^{A ***}$ Algae Red corticated foliose ^{A} Algae Red encrusting $^{M **}$ Algae filamentous ^{A} Seagrass $^{M ***}$ Porifera $^{M **}$	Algae Brown frondose ^{M} Algae Brown leathery ^{P} Algae Green articulated ^{M} Algae Green frondose ^{M} Algae Red corticated foliose $^{P ***}$ Turfing Algae ^{M} Ascidian solitary $^{M ***}$ Seagrass ^{M} Porifera ^{M}	Algae Brown frondose Algae Brown leathery Algae Red articulated Algae Red corticated foliose Algae Red encrusting Algae Red frondose Turfing Algae Sediment	
Shoalwater Islands Marine Park	Algae Brown frondose $^{A **}$ Algae Green corticated terete ^{A} Algae Red articulated $^{A ***}$ Algae Red corticated foliose ^{A} Algae Red frondose $^{S *}$ Algae filamentous ^{A} Algae frondose ^{A} Turfing Algae $^{A ***}$ Seagrass $^{S ***}$ Sediment $^{S **}$	Algae Brown frondose $^{P *}$ Algae Brown leathery ^{S} Algae Green filamentous ^{S} Algae Red articulated $^{P ***}$ Algae Red encrusting ^{P} Algae Red corticated foliose $^{P *}$ Turfing Algae $^{P **}$ Seagrass ^{S}	Algae Brown frondose $^{M *}$ Algae Brown leathery ^{S} Algae Green frondose ^{M} Algae Red articulated $^{M *}$ Algae Red encrusting ^{M} Algae frondose ^{M} Turfing Algae $^{M **}$ Sediment $^{S **}$ Porifera $^{M +}$	Algae Brown leathery Algae Red encrusting Algae Red frondose Turfing Algae Sediment

Level of ANOVA significance indicated by $*P < 0.05$, $**P < 0.01$ and $***P < 0.001$ and Mann-Whitney U test $^{+}P < 0.05$

A high degree of uniqueness for the locations in summer is illustrated by the leave-one-out allocation success rate from the CAP analysis that was >88 %. A clear separation of the locations driven by category cover types is shown in Figure 2.2 c. In this instance AB is being separated from the other locations by a stronger presence of red articulated algae, SWIMP by the presence of seagrass, PP by red corticated foliose algae and MMP by a combination of Ascidiacea, Porifera and red and green frondose algae (Figure 2.2 c).

Winter also had a strong degree of uniqueness for the locations, illustrated by the leave-one-out allocation success rate from the CAP analysis of >82 %. However, although locations were separated there was more spread for the sites within the locations, especially for PP and SWIMP Figure 2.2 d. In this case AB is separated from the other locations by a greater percentage cover of red articulated algae, PP by brown hollow algae and red corticated foliose algae, SWIMP by the presence of seagrass, red frondose algae and sediment and MMP by a combination of Porifera, and green frondose algae (Figure 2.2 d).

2.3.2 Invertebrates

2.3.2.1. Quadrats

The density of invertebrates using the quadrat method was significantly different for locations (PERMANOVA, P <0.05: Table 2.14). However there was no significant difference for fishing status. An *a posteriori* PERMANOVA for location showed no significant effects among locations, however ANOSIM showed that Parker Point was different to all the other locations (Tables 2.15 and 2.16). SIMPER revealed this difference in location could be attributed to six invertebrates with 10 significant pair-wise comparisons (1-way ANOVAs and Mann-Whitney U Tests, Table 2.17). These significant results were plotted on CAP graphs to discern any patterns being driven by them. The degree of uniqueness for the locations from the quadrat counts is illustrated by the leave-one-out allocation of success rate from the CAP analysis that was >76 %. The separation driven by the different invertebrate categories is shown in Figure 2.3 a. MMP and SWIMP tend to overlap whilst the Rottnest sites, AB and PP are clearly separated. MMP is being separated from the other sites by higher densities of Asteroidea, Ascidiacea and Zoanthidea, however there are no clear drivers for the other locations (Figure 2.3 a).

Table 2.14: PERMANOVA of the differences in the invertebrate densities for location and fishing status from the quadrat and belt transect method

	Source	*df*	MS	F	*P* (perm)
Quadrats	Location	3	1834	3.2985	0.0182*
	Fishing status (location)	3	475.87	1.2505	0.2505
	Error	10	369.45		
Belt transects	Location	3	4654.8	4.2685	0.0057**
	Fishing status (location)	3	351.09	0.4448	0.9497
	Error	10	789.38		

Data were square root transformed and analysis based on the Bray-Curtis dissimilarity measure, permutated 9999 times. Significance level is indicated by $*P < 0.05$, $**P < 0.01$ and $***P < 0.001$, *df* degrees of freedom and *MS* mean squares

Table 2.15: An *a posteriori* PERMANOVA pair-wise comparisons among locations for invertebrate density using the two estimation methods, quadrats and belt transects.

	Groups	*t*	*P* (perm)	*P* (Monte-Carlo)	No. unique values
Quadrats	A, M	1.7915		0.1288	3
	A, P	1.6661	0.0567		3489
	A, S	2.1336		0.1278	210
	M, P	2.3602	0.0609		3487
	M, S	1.3337		0.3294	210
	P, S	1.9620	0.1292		1260
Belt transects	A, M	1.2533		0.2926	0.2926
	A, P	3.5508	0.0061**		0.0086
	A, S	2.3217		0.0742	0.0742
	M, P	2.3663	0.0254*		0.0352
	M, S	1.2866		0.3355	0.3355
	P, S	2.6084	0.0401*		0.0628

Number of permutations = 9999, the Monte-Carlo *P*-value was used when there were small number of unique values. Data were square root transformed and based on the Bray-Curtis dissimilarity measure. Level of significance indicated by $*P < 0.05$, $**P < 0.01$ and $***P < 0.001$. *A* Armstrong Bay, *M* Marmion Marine Park, *P* Parker Point, *S* Shoalwater Islands Marine Park

Table 2.16: ANOSIM pair-wise comparisons among locations for invertebrate density using the two estimation methods, quadrats and belt transects.

	Groups	R	P (perm)	Actual Permutations	Number >= Observed
Quadrats	A, M	0.521	0.086	35	3
	A, P	0.444	0.008**	126	1
	A, S	0.240	0.086	35	3
	M, P	0.831	0.008**	126	1
	M, S	0.427	0.057	35	2
	P, S	0.569	0.008**	126	1
Belt transects	A, M	0.000	0.371	35	13
	A, P	0.456	0.040*	126	5
	A, S	0.167	0.200	35	7
	M, P	0.406	0.048*	126	6
	M, S	0.427	0.143	35	5
	P, S	0.194	0.183	126	23

Number of permutations = 9999. Data were square root transformed and based on the Bray-Curtis dissimilarity measure. Level of significance indicated by $*P < 0.05$, $**P < 0.01$ and $***P < 0.001$. *A* Armstrong Bay, *M* Marmion Marine Park, *P* Parker Point, *S* Shoalwater Islands Marine Park. *R* R statistic

Table 2.17: Categories identified by SIMPER as characterising the invertebrates within locations from the quadrat counts (shaded boxes) and those that distinguished between the locations (non-shaded boxes). For the pair-wise comparisons the superscript indicates in which location, Armstrong Bay (A), Parker Point (P), Marmion Marine Park (M) and Shoalwater Islands Marine Park (S), densities were highest.

	Armstrong Bay	Parker Point	Marmion Marine Park	Shoalwater Islands Marine Park
Armstrong Bay	Gastropoda Porifera			
Parker Point	Ascidiacea ^{A*} Asteroidea ^{A***} Echinoidea ^{P***}	Gastropoda Porifera		
Marmion Marine Park	Ascidiacea ^{M} Echinoidea ^{M***} Gastropoda ^{M} Porifera ^{M} Zoanthidea ^{M}	Ascidiacea ^{M**} Asteroidea ^{M***} Gastropoda ^{M} Porifera ^{M} Zoanthidea $^{M+}$	Ascidiacea Gastropoda Porifera	
Shoalwater Islands Marine Park	Ascidiacea ^{A} Porifera ^{A} Zoanthidea ^{S}	Ascidiacea ^{S} Porifera ^{S} Gastropoda ^{S} Zoanthidea ^{S*}	Ascidiacea ^{M*} Asteroidea ^{S*} Gastropoda ^{M} Porifera ^{M}	Gastropoda Zoanthidea

Level of ANOVA significance indicated by $*P < 0.05$, $**P < 0.01$ and $***P < 0.001$ and Mann-Whitney U test $^{+}P < 0.05$

2.3.2.2. Belt Transects

The density of invertebrates using the belt transect method was also significantly different for location (PERMANOVA, $P < 0.05$: Table 2.14). However again there was no significant difference for fishing status. An *a posteriori* PERMANOVA for location showed Parker Point was significantly different to all the other locations, whilst ANOSIM showed that Parker Point was only different to Armstrong Bay and Marmion Marine Park (Tables 2.15 and 2.16). SIMPER revealed this difference in locations could be attributed to six invertebrate categories with 7 significant pair-wise comparisons results (1-way ANOVAs and Mann-Whitney U Tests, Table 2.18). These significant differences were plotted on the CAP graphs so any drivers of patterns could be visualised. The degree of uniqueness for the locations from the belt transect counts was poor, illustrated by the leave-one-out allocation of success rate from the CAP analysis that was only >47 %. Locations were more overlapped than with the quadrat method and there were no clear drivers of location patterns (Figure 2.3 b).

Table 2.18: Categories identified by SIMPER as characterising the invertebrates within locations from the belt transect counts (shaded boxes) and those that distinguished between the locations (non-shaded boxes). For the pair-wise comparisons the superscript indicates in which location, Armstrong Bay (A), Parker Point (P), Marmion Marine Park (M) and Shoalwater Islands Marine Park (S) they were dominant.

	Armstrong Bay	Parker Point	Marmion Marine Park	Shoalwater Islands Marine Park
Armstrong Bay	Asteroidea Gastropoda Porifera			
Parker Point	Scleractinia ^{A} Echinoidea ^{P} Gastropoda ^{A} Porifera ^{A**}	Porifera		
Marmion Marine Park	Ascidiacea ^{M} Scleractinia ^{A} Echinoidea ^{M*} Porifera ^{A}	Ascidiacea ^{M**} Asteroidea ^{M***} Scleractinia ^{M} Gastropoda ^{M} Porifera ^{M}	Ascidiacea Gastropoda Porifera	
Shoalwater Islands Marine Park	Ascidiacea ^{A} Scleractinia ^{A***} Gastropoda ^{A} Porifera ^{A}	Ascidiacea ^{S*} Echinoidea ^{P} Gastropoda ^{S} Porifera ^{S}	Ascidiacea ^{M*} Echinoidea ^{M} Gastropoda ^{S} Porifera ^{S}	Gastropoda Porifera

Level of ANOVA significance indicated by $^{*}P < 0.05$, $^{**}P < 0.01$ and $^{***}P < 0.001$ and Mann-Whitney U test $^{+}P < 0.05$

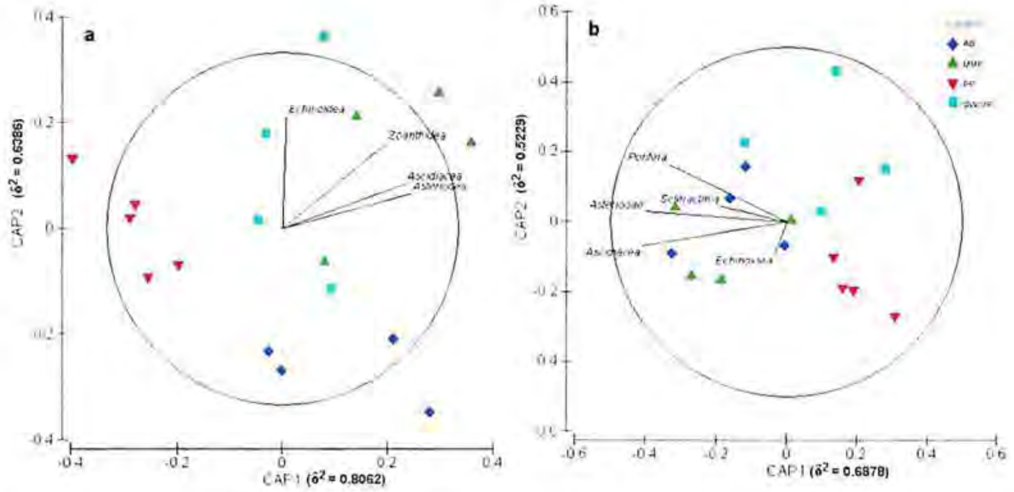

Figure 2.3: Canonical analysis of principal coordinates (CAP) ordination for locations and biplots for a) quadrats b) belt transects.

2.3.3. Kelp Holdfasts

Mean number of kelp holdfasts ranged between 3.8 and 7.5 holdfasts m^{-2} (Figure 2.4). Rottnest sites had very similar densities of kelp, these densities were generally less than that in the nearshore locations at MMP and SWIMP, however there were no significant differences for location or fishing status (PERMANOVA, $P > 0.05$) (Figure 2.3).

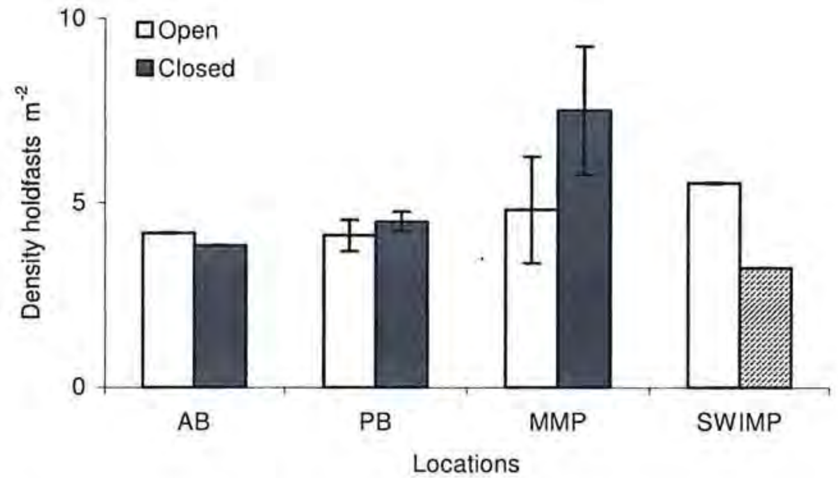

Figure 2.4: Mean density $m^{-2} \pm$ SE of kelp holdfast comparing the open to closed sites for the four locations Armstrong Bay (AB), Parker Point (PP), Marmion Marine Park (MMP) and Shoalwater Islands Marine Park (SWIMP). Note the closed site for SWIMP is the proposed sanctuary zone (hashed).

2.4. Discussion

This project has produced baseline data on the benthic communities at three marine reserves along the Swan region: in the north at Marmion Marine Park; in the south at Shoalwater Islands Marine Park; and offshore at Rottnest Island. As one of the main goals of this work was to develop a long-term monitoring program, different methods were trialled to quantify the benthic communities, including digital photoquadrat imagery and video footage of benthic cover and quadrat and belt transect counts of invertebrates. A discussion of these methods follows so that future surveys can take advantage of these recommendations and maximise the efficiency and effectiveness of their methods.

2.4.1. Method Comparisons

2.4.1.1. Photoquadrat and Video Imagery

Identifying algal specimens to species is a costly exercise, both in terms of time and effort. Often taxonomic specialists are required to identify species and specimens need to have reproductive parts for accurate identification. This requires the extensive collection of specimens, whereas identification to functional morphology level is possible from high quality digital photoquadrat and video imagery (Celliers et al. 2007). To remove substantial amounts of algae from sanctuary zones seemed contrary to the purpose of protection and was avoided by DoF for this pilot monitoring program, rather functional morphology was tested as a potential indicator of community change. Numerous studies have shown that algal functional groups can be used as an effective means for interpreting complex community patterns without having to rely on species identification (Hirst 2006;Littler and Littler 1984;Steneck and Dethier 1994;Wernberg and Goldberg 2008). A novel approach was used in this study which involved the additional clarification of adding algae phylum to the functional group, that is assigning the algae first to phylum and then to a functional group. This resulted in increasing threefold the number of classifications possible. Ultimately the removal of specimens was avoided as it may have confounded the findings of subsequent surveys, the goal of which was to find indicators affected by changes in fishing pressure.

For both methods (photoquadrat and video) multivariate analyses showed there were significant differences in the percentage cover of the flora and fauna categories for both season and location showing that both methods could be used to detect a difference. For both methods seasonal differences were characterised by algae functional groups alone (i.e. no faunal groups) with good agreement of which functional groups. Such that leathery brown algae and frondose algae was indicated by both methods as characterising summer and articulated red algae and turfing algae by both for characterising winter. Within location differences also had very good agreement among locations, with all sites for most analyses (Photoquadrat ANOSIM the exception) being shown to be significantly different to each other. The video imagery indicating in summer a slightly higher number (16) of flora and fauna categories contributed to the difference compared to the photoquadrat (12). Whilst in winter there were the same number. However it should be noted that there were more significant differences in the pair-wise comparisons from the video footage, most likely due to the greater number of samples analysed from this method. CAP analyses for both methods showed clear separation for the summer data with similar flora and fauna categories being the drivers of patterns in locations for both. For both methods, seagrass was an indicator for SWIMP, red corticated foliose algae was an indicator for PP and Ascidiacea for MMP. AB did not show this with high percentages of brown leathery algae being seen in the photoquadrats, whilst high percentages of articulated red algae were seen in the video imagery. These different results again are likely to be due to the greater number of video images analysed, compared to photoquadrat images, which would have allowed a greater chance for the smaller species to recorded. This increased sample size is also likely to explain the addition of Porifera as an indicator species for MMP, which was drawn out by the video analyses. In winter the CAP analyses showed reduced separation of the locations for the photoquadrats, whilst the video maintained a clearer separation. Similar indicator species to those in summer were seen to be driving the location patterns in the video footage in winter. Particularly the red articulated algae at AB, red corticated foliose algae at PP, seagrass and SWIMP and Porifera at MMP.

As both these methods showed quite good agreement (the video giving slightly more comprehensive information) with each other in terms of their results, consideration as to which method to use for subsequent monitoring programs would benefit from an assessment of the time and ease with which field data was collected and laboratory processing time.

In the field the heavier photoquadrat frame was far more difficult to manoeuvre especially over the heavily kelp dominated terrain and in shallower areas and where the terrain was uneven. Dive times for capturing the images was user dependent but tended to be between 100 – 180 minutes for one complete data set (e.g. open site for a location). Dive times of 180 mins significantly reduce the probability of increasing the replication of future sampling, as they are prohibitive in their duration due to increased length of surface intervals between dives. Thus, to increase replication, significantly more days in the field would be required with all the associated increased costs of boats, equipment and staff. In contrast dive times for collecting the video footage were between 30 – 40 minutes with no difference between users. Although for this method the diver was more influenced by swell and current conditions (whereas the photoquadrat frame was weighted), overall the video was more user friendly in the field. In contrast to the photoquadrats the shorter dive times for video imagery would also allow for increased sample replication without significantly increasing the number of days in the field. Time to process one image in the laboratory for either method varied according to how complex the composition of the benthos in the photograph was, but was between 5 to 15 mins for both. To process the video footage took considerably longer than the photoquadrats due to the number of images being analysed.

2.4.2. Invertebrates

The identification of invertebrates to species level not only requires a high level of taxonomic expertise, but it is also costly in terms of time and money (Wlodarska-Kowalczuk and Kedra 2007). With this in mind and the results of other studies it was decided to trial the use of higher order taxonomic levels for grouping of invertebrates (see Vanderklift et al 1998). Although direct comparisons between the two methods was not possible as they were not undertaken concurrently, indirect comparisons can be made, being mindful of the possible influence of seasonality in the results. For both methods there was a significant difference for location. However the *a posteriori* analysis of the quadrat method failed to show any differences between the sites, although ANOSIM did show that PP was different to all the other locations. The belt transect method *a posteriori* analyses also showed that PP was different to all the other locations. For both methods six of the broad invertebrate groups characterised difference between the locations. The quadrat method indicated ten of the pair-wise comparisons were significantly different, whilst the belt transect only indicated 7 were significantly different. The greatest difference between these two methods was evident in the CAP analyses. The belt transect results showed no clear indicators that could be used to separate the locations whereas the quadrat method did start to show some drivers of patterns. Asteroidea, Ascidiacea and Zoanthidea were indicators for MMP. However no other locations showed any clear drivers. The fact that there were limited results from both of the methods suggest that the sampling replication needs to be increased, for either method, and the level of taxonomy may have been too broad to identify any real indicator species.

However consideration as to which method to use for subsequent monitoring programs would benefit from an assessment of the time and ease with which field data was collected. Dive time taken to count invertebrates along the belt transect were up to 120 minutes per data set (e.g. open site for a location). Whereas for the quadrat method dives times were about half that and between 60 – 80 minutes. It would therefore be more reasonable to increase the replication of the quadrats counts.

2.4.3. Influence of Sanctuary Zones on Benthic Communities

Studies have shown that algal communities in sanctuary zones may be controlled by fishes in two ways, firstly by predation by fishes on herbivores and secondly by herbivores directly grazing on the algae (Sala and Zabala 1996). Whilst in the unprotected areas herbivores, such as fishes and urchins, in the absence of higher order predators may control the communities (Elner and Vadas 1990). For this present study all analyses undertaken showed there was no indication that protection from fishing had any influence on either the benthic communities (flora and fauna categories and kelp holdfast counts) or the invertebrates. At Rottnest Island this is most likely the result of only recent gazettal of the sanctuary zones. However, as MMP has had a sanctuary zone since 1999 it would not have been unreasonable to expect to see some differences due to fishing pressure as has been documented in other studies (see Sala 1997, Edgar and Barrett 1999;Ruitton et al. 2000). However for MMP this is not the case. It is most likely that the sanctuary zone is too small to afford protection to the predators (in terms their home range) within it so there are no cascading trophic effects. However the lack of difference between the open and the new and proposed sanctuary zones within a location does suggest that natural intrasite variability is minimal, which will make clearer any future differences due to fishing effort should they occur.

The counting of kelp holdfasts also failed to show any difference, as a result of the flow on effects of protection, for any location. This contrasts to the findings of Babcock and co-workers (1999) who found that in sanctuary zones, due to decreased grazing pressure, kelp populations increased significantly as an indirect effect of protection on the benthic habitat. However, once again, in the newer sanctuary zones (Rottnest Island) it was unlikely that any difference would have been seen due to the short amount of time of protection.

2.4.4. Conclusions and Recommendations

This study provided baseline quantification of benthic communities, flora and fauna, in three marine reserves along the Swan region. No effect of protection from fishing was observed in either the flora and fauna categories, invertebrates or kelp holdfast investigations. Some indicators (flora and fauna categories, invertebrates) of location differences were drawn from the analyses, however as this project is based on only one year of data these results should be used with caution. Most importantly it appears from this study that any shifts in the benthic communities will take longer to be seen than results for either the Western Rock Lobster or fish communities (see subsequent sections), and in this respect long-term monitoring in the vicinity of decades is probably necessary to measure change in these communities as a result of protection.

The following are a series of recommendation for future monitoring programs.

1. Continue the use of algal functional groups as future indicators under the present funding constraints as this is a more cost effective method.
2. Seasonal information should be gathered for the next 3 to 5 years to establish a baseline of seasonal variation however, after this time it may be possible to reduce sampling to just summer, with winter sampling occurring only as required.
3. Continue to collect video footage as this would allow greater sampling replication without significantly increasing the field time.

a. Increase the sampling replication by increasing the number of transects at each location and possibly the number of sites.

b. However it is suggested that the photoquadrats method be used in conjunction with the video method every 5 years to ensure that video footage continues to be the most effective method..

4. Use quadrat counts for quantifying invertebrates as this was more time efficient in the field.

    a. Increase the sampling replication, by increasing the number of transects at each location.

    b. Test the effect of increasing the level of taxonomic identification – it may be that with increased replication the present broad level of identification may be sufficient.

    c. Include artificial collectors to sample for nocturnal/cryptic species.

5. Future studies would benefit from the inclusion of environmental data, including water temperature, water quality and hydrodynamic information.

3. Western Rock Lobster

Jason How and Simon de Lestang

3.1 Introduction

The western rock lobster (*Panulirus cygnus*) forms the basis of the largest single species fishery in Australia worth between A$250 – 400 million annually (Caputi et. al 2008). The species is also captured recreationally through both potting and diving and equates to approximately 5% of the total catch. Most recreational effort is focused in the first few months of the rock lobster fishing season around the Perth Metropolitan region (Melville-Smith & Anderton 2000).

As rock lobsters are abundant in the Swan region and they exhibit known responses to environmental changes (Melville-Smith and de Lestang, 2006), these animals' make good candidates as biological indicators. Furthermore, understanding the dynamics of these populations' aids in their management, and that of associated faunal communities.

This project aims to provide both a baseline of biological information for Western Rock Lobster in the Swan Catchment Region as well as a cost-effective and efficient sampling protocol for the continued assessment and recording of these biological parameters into the future. In the long-term the work aims to produce a long-term time series that will be a robust indicator of the relative "health" of the Swan Catchment ecosystem into the future.

3.2 Methods

Within the three marine reserves (1.8.2 – 4) four locations were sampled; Marmion Marine Park – "the Lumps" (MMP), Shoalwater Islands Marine Park – proposed Penguin Island Sanctuary zone (SWIMP), Rottnest Island Marine Reserve – Armstrong Bay (AB) and Rottnest Island – Parker Point (PP). At each location, three sites inside the closed zone and three sites in the open fishing areas adjacent to the closed zones were sampled using dive and pot based survey techniques. At each location, sites were chosen based on their similarity in a range of abiotic (e.g. water depth, substrate structure and distance offshore) and biotic factors (dominating floral community).

3.2.1 Pot Based Surveys

Three pot types were used at each location so as to sample as wide a size-range of the lobster population as possible. The pots used were a combination of standard batten commercial pots, standard batten commercial pots with all gaps between the battens reduced to < 15 mm (meshed commercial) (Figure 3.1) or standard batten recreational pots. All pots had their escape gaps covered.

At each site within a location at least one of each type of lobster pot was fished. Over the course of the study each pot type fished each site at least three times, providing a standard number of pot days for each pot type within a site. The pots were each baited with blue mackerel, *Scomber australasicus* (~1 kg) and set for ~ 24 h at the base of ledges throughout the study region. The GPS position of each pot set was noted.

A recorded was kept of all lobsters caught including the pot type they were captured in, their carapace length (CL), sex, the presence of any appendage damage and their reproductive state (e.g. the presence of ovigerous setae, spermatophores or eggs). All lobsters captured with all of their appendages intact were tagged with a t-bar spaghetti tag before being released back near their capture location. A note was kept of all pots that contained or appeared to have been attacked by octopus or fish. Since the presence of predators within a pot reduces the catch rate of lobsters (Brock et al. 2006), these pots have been removed from all catch rate analysis.

Figure 3.1: Examples of the meshed commercial (left) and standard commercial (right) batten pots

3.2.2 Diving Based Surveys

Underwater visual census (UVC) of lobster size and abundances were undertaken using SCUBA. At each site (six sites per location), three 30 m long x 2 m wide transects were surveyed covering "good lobster habitat" (transect area $60m^2$) (Figure 3.2a). Good lobster habitat was considered to be areas of high relief limestone reef with associated algal growth.

Figure 3.2: a) Image of diver surveying transects along "good lobster habitat" with a tape measure to his right and b) underwater measurement of lobster after *in situ* size estimation for validation.

The two-diver teams consisted of experienced divers who were familiar with the diurnal microhabitat occupied by lobsters, and were experienced in recreational diving for lobsters. One diver deployed a 30 m long tape along suitable habitat, following ledges and/or crevices, while a second diver followed the tape recording lobster abundance and sizes within 1 m of the tape (Figure 3.2a). The second diver also attempted to catch (via a lobster loop) lobsters once their carapace length had been estimated. All diver-captured lobsters were measured underwater using callipers to the nearest millimetre (Figure 3.2b) before being placed in a catch bag to be returned to the research vessel. Once onboard lobsters again had their details recorded before being tagged and returned to the water alive.

3.2.3 Analysis

All data collected was entered onto Microsoft Access for storage. Once extracted the data was manipulated and analysed using R (R Development Core Team 2008))

ANOVAs were used to examine the differences in the abundance of legally retainable (legal), undersize and all lobsters between marine reserve location, activity zoning and pot type. Tukey HSD post hoc tests were used to assess all significant pair-wise differences identified by ANOVA. All abundance data, from both UVC and potting, required $\sqrt[4]{}$ transformation of the data to remove skewness (Clark and Warwick 2001). All abundance data has been presented in its transformed form with 95% confidence intervals.

As Shoalwater was not gazetted as a marine reserve, it was removed from ANOVAs examining the effect of zoning. It is however presented in figures showing either CPUE or relative abundances between the open and proposed closure.

Legal lobsters were classified as such based on the current legally retainable catch requirements of the fishery in January (Caputi et al., in press). This includes lobsters with CLs $\geq$ 77 mm, females with CLs < 115 and not in a setose condition (de Lestang and Melville-Smith, 2006). For underwater visual census, actual carapace length (CL), sex and reproductive state could only be determined from captured individuals. As such, those individuals who were surveyed and not captured were assumed to be legal if the estimated size was $\geq$ 77mm.

Comparisons of the size composition of lobsters between methods in each location / activity zone were done using Kolmogorov – Smirnov tests. Significance levels of these tests were adjusted using a *Bonferroni correction*, accounting for multiple pair-wise comparisons.

3.3 Results

3.3.1 Influence of Survey Method on Size Distribution

3.3.1.1 Potting

Pot captured lobsters ranged in size from 44.7 to 116.2 mm CL, with most lobsters having CLs between 60 and 95 mm (Figure 3.3). The meshed commercial pots captured the smallest lobster (44.7 mm CL) while the largest lobster was captured in the standard commercial pots (116.2 mm CL).

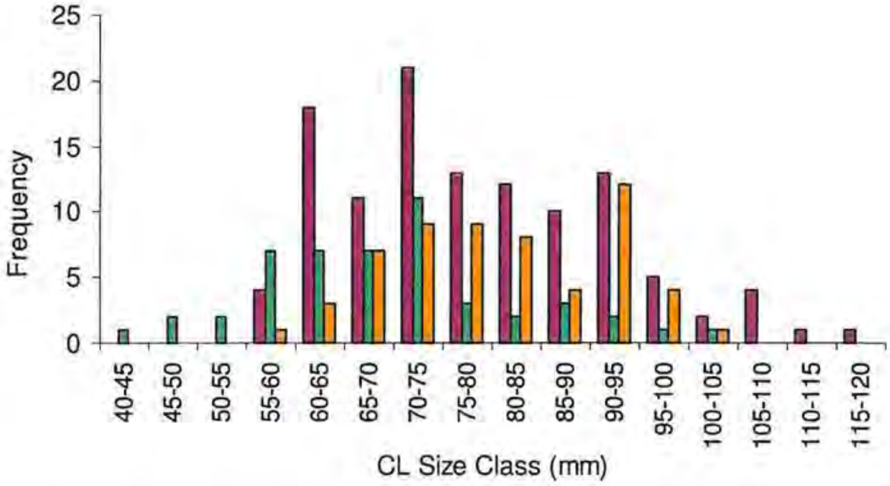

Figure 3.3: Size frequency distribution of lobsters captured using standard commercial (maroon), meshed commercial (green) and recreational (orange) pots.

The overall size composition of lobsters captured by each pot-type differed significantly ($p < 0.001$) (Table 3.1), with the meshed commercial pots catching a significantly greater proportion of smaller lobsters than either the standard commercial or recreational pots (Figure 3.3). No significant difference in the size composition of lobsters was recorded between the standard commercial and recreational lobsters pots (Table 3.1).

Table 3.1: Pairwise comparisons of the size-composition of lobsters caught in the three pot types (standard commercial – Com; meshed commercial – Mesh; recreational – Rec). (*Bonferroni* adjusted significance level $p < 0.017$; bold indicates a significant difference)

Pot Type	Pot Type	Test statistic (D)	*p*-value
Com	Mesh	0.2876	**0.003**
Com	Rec	0.1385	0.430
Mesh	Rec	0.3893	**<0.001**

3.3.1.2 Diving (Underwater Visual Census)

3.3.1.2.1 Visual Census Size Estimation

Nineteen lobsters were caught and measured during diving surveys to examine the accuracy of visual size estimation. These lobsters ranged in size from 61 – 148mm CL. The largest lobster was removed from this comparison as it was atypical in size and had the potential to disproportionately bias the analysis. In a linear regression, its far larger size would have resulted in this point having a far greater weighting than the rest of the data set, hence its removal from subsequent analysis.

The estimated CLs of lobster from UVC did not significantly differ from the respective measured CLs (intercept, $p = 0.12$; slope, $p = 0.11$). Moreover the intercept of the two lines was at 75 mm CL, highlighting the accuracy of size estimates at or around legal minimum size (77 mm CL). However it should be noted that the small number of comparisons used in this analysis (n= 18) limited the power of this test. A greater number of observations will be collected in the future.

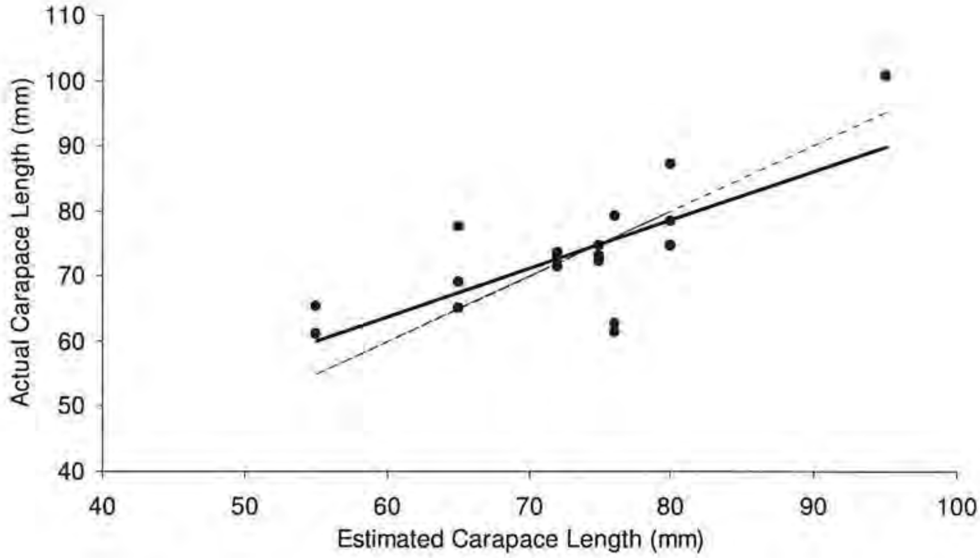

Figure 3.4: Estimated and actual carapace lengths of lobsters captured during underwater visual census (UVC), showing their relationship to each other (solid) and a 1:1 relationship (dotted) where estimate CL equals actual CL.

3.3.1.3 Potting and Diving

The similarity of the size compositions derived from the standard commercial and recreational pots (Section 3.3.1.1), allowed these data sets to be combined before comparing size composition data derived from pots with that from UVC.

Under water visual census (UVC) recorded the smallest (20 mm CL) and largest 148 mm CL) lobsters captured during the study (Figure 3.5). The size composition of lobsters derived from UVC was very similar to that produced by the meshed commercial pots (i.e. a greater proportion of smaller lobsters), but differed significantly ($p < 0.001$) from that derived from the combined standard commercial and recreational pots (Table 3.2).

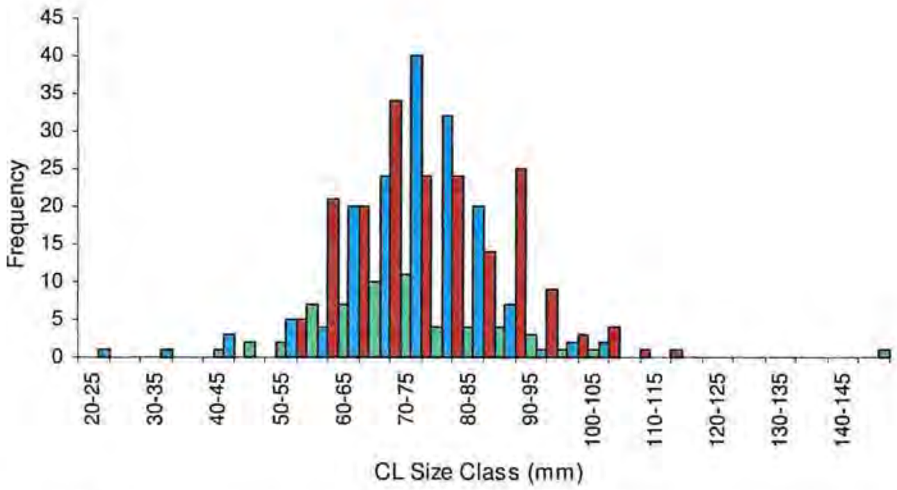

Figure 3.5: Comparative size frequency distributions through potting (commercial and recreational combined – maroon; mesh – green) and UVC (blue)

Table 3.2: Pairwise comparisons of the size-composition of lobsters caught by pots (standard commercial and recreational combined "Com & Rec"; meshed commercial – "Mesh") and recorded by UVC. (*Bonferroni* adjusted significance level $p < 0.025$; bold indicates a significant difference)

Method	Method	Test statistic (D)	p-value
UVC	Com & Rec	0.3945	**<0.001**
UVC	Mesh	0.2299	0.2766

3.3.2 Effect of Zoning on Lobster Abundance and Size Distribution

3.3.2.1 Potting – Catch Rates

The catch rate of all lobsters, legal lobsters and undersize lobsters all differed significantly between marine parks and activity zoning but not between pot-types (Table 3.3). Of the significant factors, presence of marine reserve accounted most for the variability in lobster catch rates (~22%), while activity zone accounted for the least (~7%).

Table 3.3: Influence of location zoning and pot type on lobster abundance. Sum of squares and p values (bold indicates significant effect).

Factor	Df	All lobster	Legal lobster	Undersize lobster
Marine Reserve (MR)	2	5.7465	3.3159	1.7920
		$p < 0.001$	**$p = 0.005$**	$p = 0.09$
Zone (Z)	1	1.8433	2.9059	1.6464
		$p = 0.01$	**$p = 0.002$**	**$p = 0.03$**
Pot type (P)	2	1.0944	0.9912	1.1752
		$p = 0.16$	$p = 0.18$	$p = 0.19$
MR x Z	2	1.1777	0.3098	1.9236
		$p = 0.14$	$p = 0.58$	$p = 0.07$
MR x P	4	1.5432	0.4426	1.3130
		$p = 0.26$	$p = 0.81$	$p = 0.44$
Z x P	2	0.2814	0.4753	0.2121
		$p = 0.61$	$p = 0.43$	$p = 0.74$
MR x Z x P	4	0.6784	0.5544	1.0223
		$p = 0.66$	$p = 0.74$	$p = 0.57$
Residuals	47	13.2567	13.1752	16.2237

Both marine reserves at Rottnest Island had significantly higher catch rates of all lobsters than the coastal marine parks of Marmion, while for legally retainable lobsters, Parker Point had significantly higher catch rates than any of the other marine parks. Armstrong Bay produced significantly higher catch rates of undersized lobsters than Marmion, but not Parker Point. Although greater numbers of undersize lobster were recorded in Parker Point closed zone than fished area, this difference was not significant.

For all size groups (all lobsters, legally retainable and undersized), there were significantly higher catch rates in the closed fishing zones compared to the open zones (Figures 3.6 – 3.8). Individually, Parker Point was the only marine reserve to show a significant difference the abundance of legal lobsters between activity zone, i.e. between closed and open fishing zones (Figure 3.7).

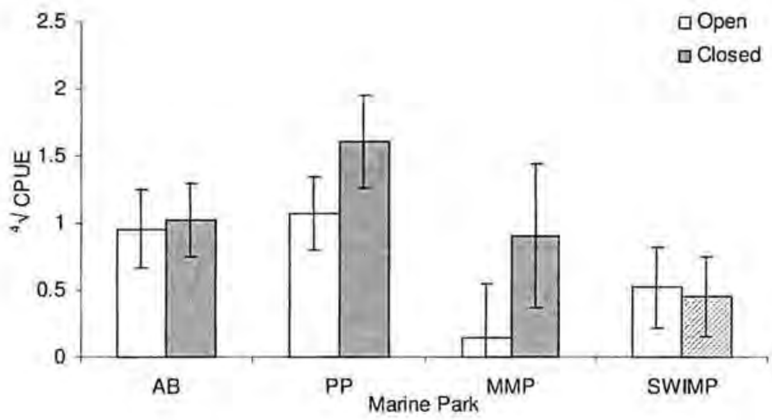

Figure 3.6: Forth-root transformed mean catch rate (±95 CI) of all lobsters in open and closed fishing zones in the four marine reserves. SWIMP closed zone (hashed) is a proposed closure

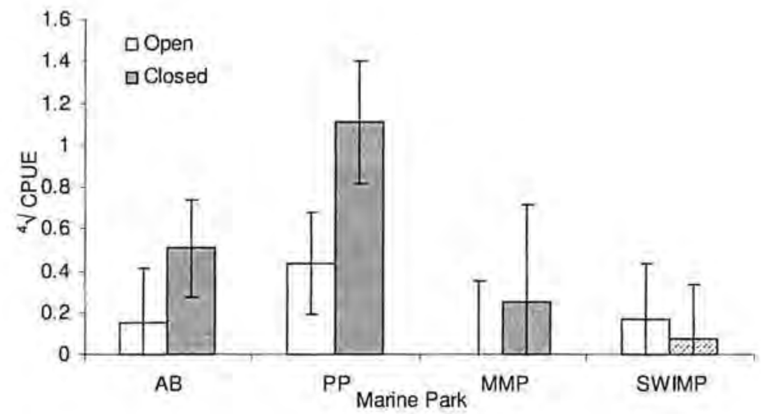

Figure 3.7: Forth-root transformed mean catch rate (±95 CI) of legal lobsters in open and closed fishing zones in the four marine reserves. SWIMP closed zone (hashed) is a proposed closure.

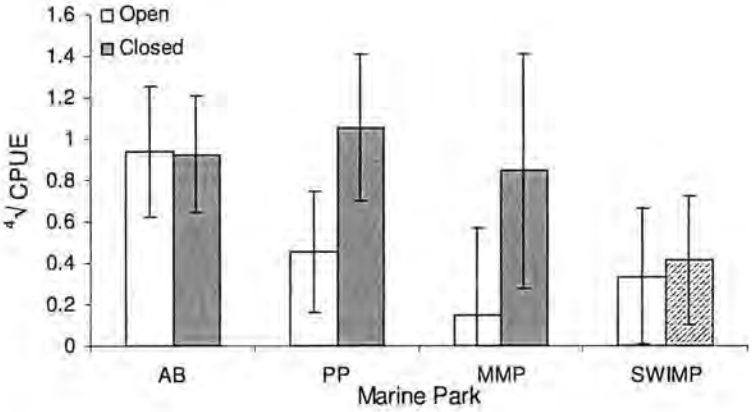

Figure 3.8: Forth-root transformed mean catch rate (±95 CI) of undersize lobsters in open and closed fishing zones in the four marine reserves. SWIMP closed zone (hashed) is a proposed closure

3.3.2.2 Potting – Size Compositions

Comparisons of the size frequency distributions of paired open and closed sites at each marine reserve indicated that activity zoning did not significantly effect size composition (Table 3.4).

Table 3.4: Paired size frequency compositions of lobster in each marine park's open and closed* fishing zones. (*Bonferroni* adjusted significance level $p < 0.0125$; bold if significant; * Shoalwater closed region is only proposed)

Location / zoning	Location / zoning	Test statistic	*p*-value
Armstrong Open	Armstrong Closed	0.2759	0.087
Parker Point Open	Parker Point Closed	0.1348	0.80
Marmion Open	Marmion Closed	1	0.29
Shoalwater Open	Shoalwater Closed	0.5833	0.14

The data for each Marine Park was therefore pooled prior to comparing the size compositions between the four Marine Parks. The size composition of lobster at Parker Point was significantly larger than at Armstrong Bay and Shoalwater marine parks, and similar to that at Marmion Marine Park (Table 3.5 and Figure 3.9).

Table 3.5: Size frequency compositions of lobster from the four marine parks (*Bonferroni* adjusted significance level p < 0.008; bold if significant)

Location		Test statistic	*p*-value
Armstrong	Parker Point	0.412	**<0.0001**
Armstrong	Marmion	0.2159	0.9227
Armstrong	Shoalwater	0.2	0.7176
Parker point	Marmion	0.5649	0.02879
Parker point	Shoalwater	0.6031	**0.0002**
Marmion	Shoalwater	0.2143	0.9829

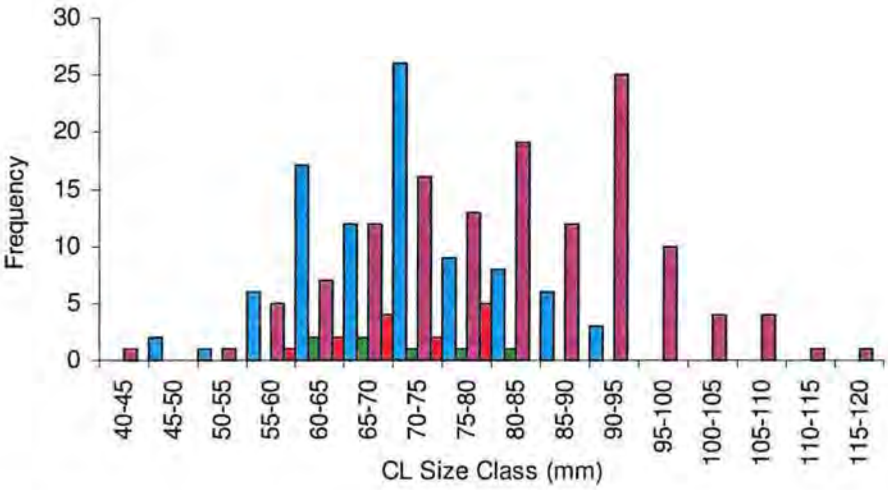

Figure 3.9: Size frequency distribution for lobsters captured by potting at Parker Point (maroon), Armstrong Bay (blue), Marmion (green) and Shoalwater (red) marine parks.

3.3.2.3 Diving Abundance (Underwater Visual Census)

ANOVA showed that, after transformation, both marine park location and zoning (open or closed to fishing) did not significantly ($p > 0.05$) affect the abundance of undersize, legal or all lobsters (Table 3.6; Figure 3.10 – 3.12).

Table 3.6: Influence of location and activity zoning on the transformed abundance of lobster. Sum of squares and p values (bold indicates significant effect).

Factor	Df	All lobster	Legal lobster	Undersize lobster
Marine Reserve (MR)	2	0.5819	1.1453	2.0478
		p = 0.56	p =0.17	p = 0.11
Zone (Z)	1	0.1398	0.6727	0.0049
		p = 0.60	p = 0.15	p = 0.92
MR x Z	2	0.4473	0.1727	0.8278
		p = 0.63	p =0.75	p = 0.40
Residuals	35	17.0112	10.8804	15.2028

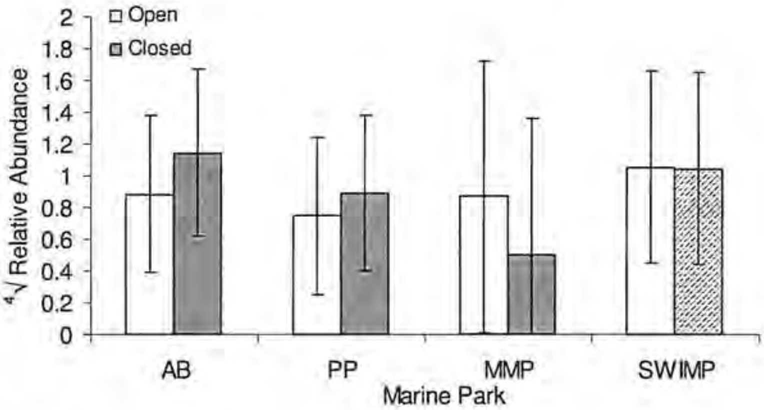

Figure 3.10: Transformed mean (±95 CI) abundance of all lobsters in closed and open fishing areas of the four marine parks. SWIMP closed zone (hashed) is a proposed closure

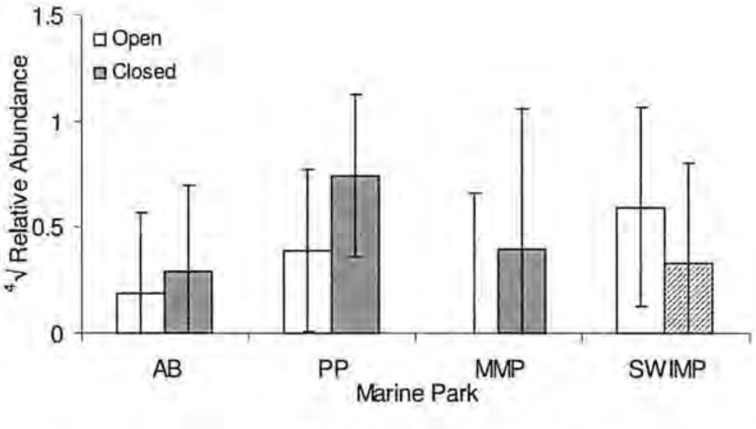

Figure 3.11: Transformed mean (±95 CI) abundance of legal lobsters in closed and open fishing areas of the four marine parks. SWIMP closed zone (hashed) is a proposed closure.

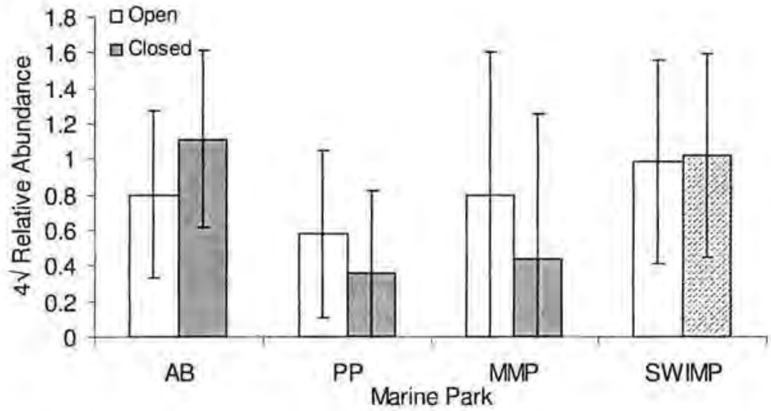

Figure 3.12: Transformed mean (±95 CI) abundance of undersize lobsters in closed and open fishing areas of the four marine parks. SWIMP closed zone (hashed) is a proposed closure.

3.3.2.4 UVC Size Compositions

Comparisons of the size frequency distributions of paired open and closed sites showed that there was no significant difference for any of the locations (Table 3.7).

Table 3.7: Paired size frequency compositions of lobster in each marine park's open and closed* fishing zones (*Bonferroni* adjusted significance level $p < 0.0125$; bold if significant, * Shoalwater closed region is only proposed).

Location / zoning	Location / zoning	Test statistic	*p*-value
Armstrong Open	Armstrong Closed	0.3841	0.03036
Parker Point Open	Parker Point Closed	0.3765	0.2239
Marmion Open	Marmion Closed	0.4	0.8692
Shoalwater Open	Shoalwater Closed	0.3537	0.04706

The data for each Marine Park was therefore pooled to test for differences in size composition between the four Marine Parks. The size composition of lobster at Parker Point was significantly larger than at Armstrong and Shoalwater marine parks, but not different from Marmion Marine Park (Table 3.5 and Figure 3.9).

Table 3.8: Size frequency compositions of lobster from the four marine parks (*Bonferroni* adjusted significance level p < 0.008; bold if significant)

Location / zoning	Location / zoning	Test statistic	*p*-value
Armstrong	Parker Point	0.4597	**0.000267**
Armstrong	Marmion	0.4875	0.04775
Armstrong	Shoalwater	0.2543	0.03875
Parker point	Marmion	0.2465	0.7867
Parker point	Shoalwater	0.4167	**0.001426**
Marmion	Shoalwater	0.4444	0.09083

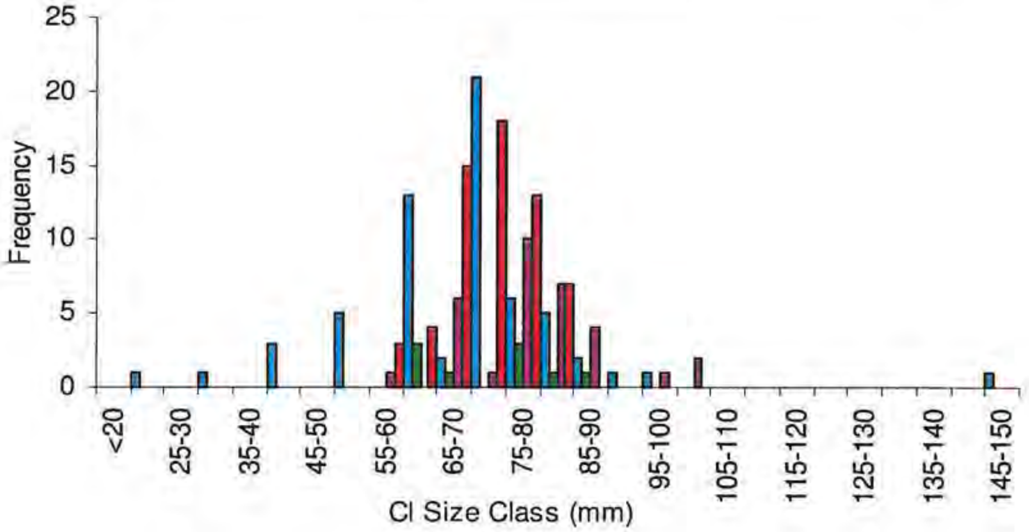

Figure 3.13: Size frequency distribution of lobsters recorded by UVC at Parker Point (maroon), Armstrong Bay (blue), Marmion (green) and Shoalwater (red) marine parks.

3.3.3 Tagging

At total of 367 lobster were tagged and released throughout the study, with 98% of these being captured by potting

Table 3.9: Number of *P. cygnus* tagged at each marine reserve / location for the two sampling season.

Marine Reserve / Location	Potting 2007	Potting 2008	Diving 2007	Diving 2008	Total
Rottnest Island – Parker Point	44	113		5	162
Rottnest Island – Armstrong Bay	97	76		2	175
Marmion	17			1	18
Shoalwater Islands	3	8	1		12
Total	161	197	1	8	367

3.3.3.1 Recapture Rate

Fourteen of the 367 tagged lobsters released as part of this study were recaptured (3.8%). All recaptures were from lobsters caught by potting in summer 2008 at the two Rottnest Marine Parks, with the vast majority (71%) being recaptured as part of this sampling protocol. All other recaptures were from recreational fishers and all from the northern side of Rottnest (near Armstrong Bay).

3.3.3.2 Movement

Four individuals were recaptured a day after tagging. They were again released and not included in all movement and growth analysis. The remaining nine recaptures had an average time at liberty of 162.5 days (range 132 – 175 days). All recaptures were recorded at the site of release, with no animals released in the closed fishing zone being caught in the open zone and *vica versa*.

3.3.3.3 Growth

Since four lobster recaptures came from recreational fishers who didn't supply carapace length, growth data was only available for six individuals. Average growth was 2.6mm (± 0.7 SE) over the average period of 165.5 days. Growth ranged from -0.2 – 4.9mm, with the largest growth increment being from the smallest recaptured individual (Figure 3.14).

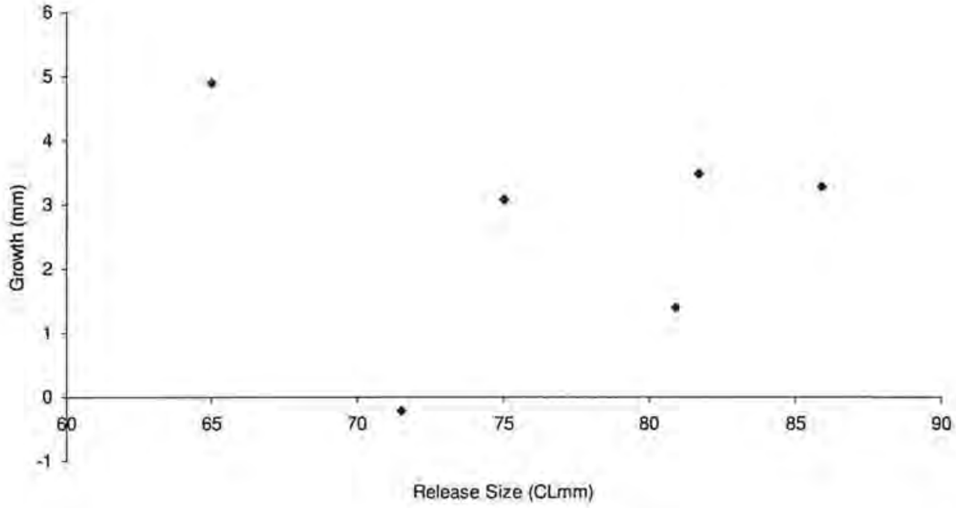

Figure 3.14: Release carapace length (mm) and growth increment for the six lobsters with recapture size and time at liberty greater than one day.

3.3.4 Size at Maturity

All mature lobsters were captured at the two marine parks at Rottnest Islands. As merus length was not measured (Melville-Smith & de Lestang 2006), it was not possible to assess the reproductive maturity of males based on this morphological technique. Maturity status in female lobsters on the other hand is easy to determine visually, being based on the presence of eggs or a spermatophoric mass (Melville-Smith & de Lestang 2006). Female size at maturity, based on data combined for the two marine parks at Rottnest Island, was CL_{50} = 79.6mm (±0.75 SE) and CL_{95} = 87.8mm (±2.0821 SE) (Figure 3.15).

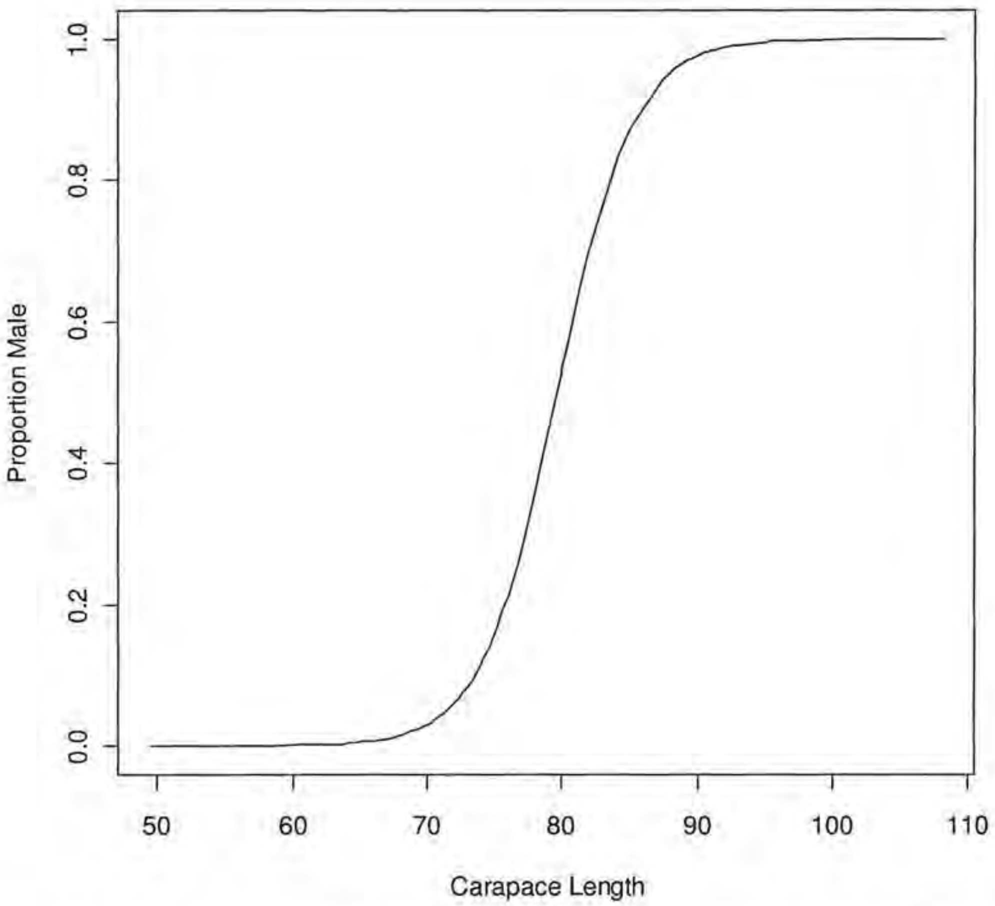

Figure 3.15: Size at maturity for female lobsters at the two marine parks at Rottnest.

3.4 Discussion

3.4.1. Method Comparison

This project has produced preliminary baseline data on western rock lobster relevant for their use as a biological indicator throughout the Swan Catchment region, both in the northern and southern regions of coastal waters and offshore at Rottnest Island. Previous work on Western Rock Lobster has highlighted how sensitive these animals are to fine scale changes in the marine environment. Significant changes in basic life parameters have been exhibited by this species in response to small-scale changes in food abundance (Edgar, 1990), fishing pressure (Babcock et al., 2007) and water temperature (Melville-Smith and de Lestang, 2006). For example, Melville-Smith and de Lestang (2006) showed that a change in water temperature of only 0.3 °C can alter size at maturity by over 4 mm CL. The fact that western rock lobsters are abundant throughout the Swan Catchment region and are relative easy of monitor, makes this species an ideal biological indicator for the ecosystem health of this region.

Although it is far too early to use the results of this study as a proxy for change in the marine environment, the continued collection of these data and the production of indices such as growth rates and size at maturity will provide a good measure of environmental change in the future.

This project has also highlighted the importance of a multi-disciplinary approach. The combination a potting program, utilising several pot types, with underwater visual census (UVC), provided a far more comprehensive assessment of the lobster population, which is essential when using a species such as this as a biological indicator or when assessing the effectiveness of closures in marine parks.

Potting enables the capture of large numbers of lobsters quickly and effectively, essential for long term low cost monitoring programs. I t also enables size, sex, and reproductive state to be accurately determined. This is a considerable improvement of traditional studies that solely utilise UVC data (Kelly et. al. 2000, Cox and Hunt 2005, Babcock et al 2007, Wynne and Côté 2007), by removing uncertainty associated with size estimations or estimates of sex or reproductive state. Accurate sex and reproductive state information for lobsters is particularly important when utilising size at maturity estimates as a biological indicator for the monitoring of environmental changes. It is only possible through the capture of lobster to accurately assess their sex and reproductive state and ascribe them to either legally retainable catch or individuals that must be returned. As legally retainable catch is the portion of the population upon which fishing mortality is focused, accurately ascribing a lobster as legal or not will influence the validity of assessments for marine reserves.

The capture of lobsters also enables tag and release programs to be conducted. Such programs provide valuable information as to the movement patterns of lobsters and their growth rates. Movement patterns of target species is pivotal to the effectiveness of marine park closure design (Kramer and Chapman 1999). Recaptures of tagged lobsters can provide an indication as to the effectiveness of reserve size in protecting targeted lobsters as well as examining possible spillover effects as densities within the closure increase.

This study has shown that lobster pots design significantly influences the proportion of the population available for capture, thus enabling the targeting of certain sizes of lobsters. For this reason, escape gaps were incorporated into western rock lobster pots in 1966 to reduce the capture of undersize lobsters. While there was no difference in the catchability of lobsters between the three pot types used in this survey (Table 3.3), they did capture a different size spectra of lobsters (Table 3.1). Intentional modifications to the mesh pot yielded greater catches of smaller lobster than occurred in either the recreational or commercial batten pots. Therefore, through using multiple pot types, one of which was specifically designed to capture the lower end of the size spectra, produced a more representative size composition of the population.

Underwater visual census is the common method used to assess lobster abundances in coastal marine reserves (Kelly et. al. 2000, Cox and Hunt 2005), particularly in Western Australia (Babcock et. al. 2007, MacArthur in press). With all UVC, size estimation accuracy must be assessed. In this study estimation of lobster sizes was relatively accuracy, especially around legal size (Figure 3.4). Accurate identification of legal and undersize lobsters is very important when assessing the impact of fishing pressure as it primarily influences legal size individuals.

Surveying lobsters via UVC removes the catchability-based biases that plague all pot based surveys, thus enabling the size distribution of the population to be better estimated. While the comparison of the size distributions recorded by UVC and small mesh potting in this study showed that there wasn't a significant difference (Table 3.2) it should be noted that the smallest (20 mm CL) and largest (148 mm CL) lobsters were recorded via UVC. This is an important point especially with regard to long-term monitoring of marine reserves. Large, especially male lobsters are unlikely to be captured by conventional lobster pots (Department of Fisheries unpublished data). The removal of fishing mortality from a population, as provided by marine reserves, is likely to result in a dramatic increase in the abundance of very large lobster (Babcock et al., 2007). Without incorporating UVC in a monitoring program, the presence of these large lobsters, and therefore of a significant amount of lobster biomass, would otherwise go unnoticed.

A multi-disciplinary monitoring program, including both potting and UVC, to assess lobster sizes and abundances within marine parks provides greater strength to interpretations of results. It enables detailed information to be collected through a variety of pot types, while capturing a complete size range through UVC. Through the use of two independent methodologies, concurrence in data trends adds greater strength to any conclusions that are drawn.

3.4.2 Influence of Marine Reserves on Lobster Populations

3.4.2.1 Marine Reserve Location

Both marine reserve location and management zoning were significant factors in the analysis of marine reserve functioning in terms of lobsters (Table 3.3). Generally, more lobsters were recorded at the Rottnest Island Marine Reserves (Armstrong Bay and Parker Point) than both of the coastal sites. This is likely to be due, at least in part, to the greater available habitat around Rottnest and the closer proximity of deeper "less exploitable" lobster grounds. The limestone reef areas surrounding Rottnest Island, which are the dominant habitat of this species, (Caputi et al., 2008) are far more expansive than the more sporadic reefs found throughout Marmion and Shoalwater Island Marine Reserves. Other factors could also have led to these differences in abundance, including marine reserve size (Table 1.1), location (Figure 1.1) time since protection (Table 1.1) and benthic communities (Section 2.4).

3.4.2.2 Marine Reserve Zoning

There have been numerous studies that have shown increases in size and abundances of a variety of marine taxa inside closed zones compared to open areas (review by Halpern 2003), and for lobster in particular (Kelly et. al. 2000, Cox and Hunt 2005), with one previous study being conducted within the Perth Metropolitan Region (Babcock et al 2007). It is therefore not surprising that there was a significant zoning effect from potting data in this study, with closed zones containing more lobsters than open zones. This difference however was only significant in an overall comparison and generally not within individual marine parks. The only significantly within marine park comparison of lobster abundance occurred at Parker Point for the legal-sized proportion of the population, which had more legal lobsters in the closed zone than the open zone. The lack of more paired comparisons was not unexpected due to the high level of within-site variation recorded, possibly the result of a lack of samples being collected at each site (generally 6 pots). As such future sampling within these regions will be expanded to better account for within site variation.

This problem of within-site variability was further highlighted in the UVC data (Table 3.6). Only three transects were conducted in each site and this lack of replication combined with the variability in habitat resulted in very large amounts of variation in the abundance estimates within each site. Lobsters are gregarious in nature, and therefore have a patchy distribution. Increased replication of UVC transects would reduce variation associated with the heterogeneous lobster distribution and reveal any zoning or marine reserve location effects.

Shoalwater Islands Marine Reserve has yet to be gazetted and therefore there aren't "closed" zones. Without differential fishing pressure it is unlikely that there would be difference between "zones". However, there is very little difference in the undersize lobsters surveyed in either potting or diving between "zones". This seems to indicate that lobster habitat between the two "zones" is comparable, as any differential fishing pressure would not impact on undersize lobsters. This is important when assessing impacts of marine park zoning, such that comparisons are made based on the effect of zoning without influence from differential habitat quality.

The Rottnest Islands Marine Reserves were only recently gazetted (July 2007). Surveys done in January 2008, while 6 months post gazetting, only equate to 2 months of fishing pressure, as the western rock lobster season commences on November 15. This two-month period receives the highest level of fishing pressure each year (Melville-Smith & Anderton 2000), which was highlighted by the fact that we received returns of tagged lobsters from recreational fishers in January. Despite this period of high pressure, it is still unlikely that there would be large differences between zoning, with the recreational catch in the entire southern zone (C Zone) being approximately 300 tonnes (2005/06 season; adapted from Caputi et al 2007). This would account for a relatively small extractive pressure around Rottnest Island. This study was also conducted just after the annual migration of pre-adult "white" lobsters, which could have re-populated much of the Rottnest Islands marine parks open fished areas.

The only significantly different paired comparison of lobster relative abundance was for legally retainable lobster at Parker Point. This is likely to reflective of local environmental conditions rather that differential fishing pressure resulting from management zoning. The Parker Point Marine Reserve is on the southern side of the island (Figure 1.1), and as such is affected by wind and swell from the prevailing southwesterly winds. The closed zone of Parker Point is an area that experiences large swells, often making the area inaccessible to lobster fishing.

Data captured from the Recreational Angler Program (RAP), (see Attachment) provided an indication of the differential fishing effort distribution for lobsters around Rottnest Islands. While the data is from only 6 fishers who reported lobsters in their catch returns, and as such should be interpreted with caution, it does show that there is considerably more effort (days fished) on the northern side of the island, with the effort around the Parker Point closure being the lowest of the three blocks where fishing was reported. There was no effort reported for block BN61, but given the small amount of coastline that is contained within the block, it is not unexpected (Figure 3.16).

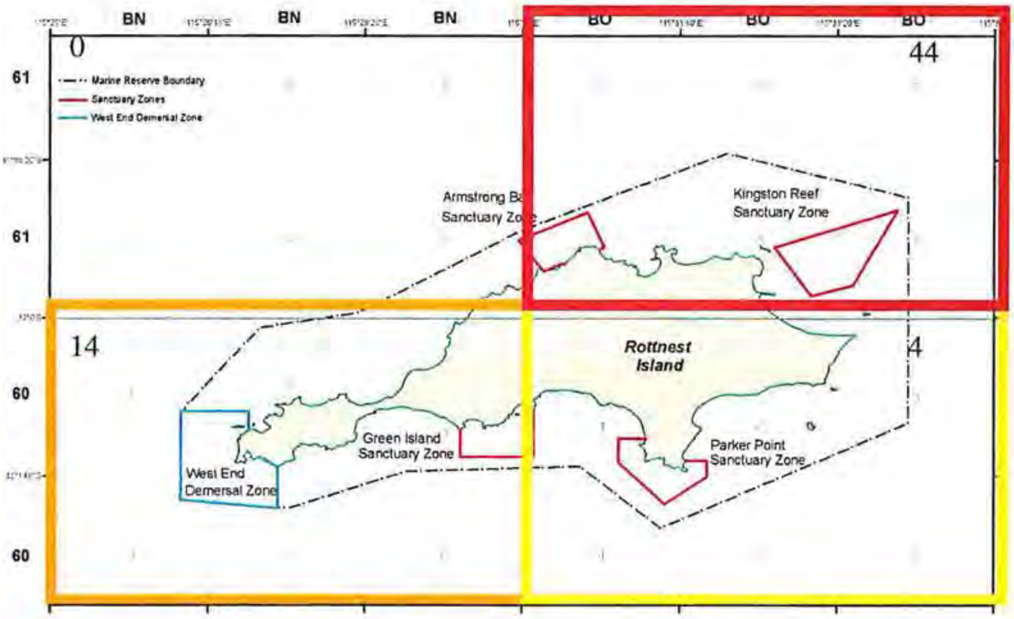

Figure 3.16: Fishing effort (fisher days) for western rock lobster in waters around Rottnest Island (Adapted from RAP attachment); white 0, yellow 1 – 10, orange 11 – 20 and red >20 fisher days. Numbers in top left corner of each block represent the number of fisher days per block.

These results highlight the importance of a baseline survey against which the effectiveness of a closure can be assessed. Singular surveys some time after protection may interpret results as an effect of zoning, without accounting for historical fishing pressure differences, independent of zoning. The higher number of undersize lobster in the Parker Point closed zone (Figure 3.8) may also indicate that it is a more preferable lobster habitat. This would further compound any results from future studies without baseline levels against which to compare.

Marmion Marine Reserve has been established for over 20 years and would be expected to show an effect of protection (Halpern and Warner 2002, Russ and Alcala 2004). There is an indication that there are more legal sized animals inside the closed area than the adjacent open area (Figure 3.7), though it isn't significant, possibly due to the large within-site variations. Another reason for the lack of difference may be the size of the closed area or its configuration. Lobsters have been shown to have variable foraging distances with foraging distances up to 800 m in a night, with most ranging between 70 – 585 m (Jernakoff et al. 1987). More recent studies show movements of within 60 m from the reef edge into flat macro-algal pavement or seagrass meadows (MacArthur et al in press). Some of these studies show movements that would place lobsters that reside with the closed areas of the Marmion Marine Park outside the boundaries of the closed zone, if they were to move perpendicular to the reef edge. While the closure is a kilometre in length, in places reef is only around 50 m from the zone boundary, making cross boarder foraging movements likely. A lack of lobster size and abundance has been noted in other small protected areas, due to the probability of lobster moving outside them being high (Cox and Hunt 2005).

Another notable issue associated with Marmion Marine Reserve is illegal fishing. The effectiveness of spatial closures is often driven by the level of compliance (Little et al. 2005). Illegal potting was recorded (and reported) within the closed zone of the marine reserve during this study. If this is a continual problem, then it may also account for similarity in lobster density between open and closed areas within this marine reserve.

3.4.2.3 Tagging and Biological Information

This study tagged and released over 350 lobsters, which and enabled a number of biological and ecological parameters to be measured. The fourteen recaptures of tagged lobster were all returned from the original site of release. This indicates that for the almost six month liberty, lobsters were relatively sedentary. Lobsters are known to undertake large migrations during the "whites" phase of their life cycle, moving over 200 kilometres, with the majority of movements being within 10 km of their release site (Chubb et al. 1999). However, recent acoustic tracking studies have shown that not all "whites" undertake long-range migrations and many actually remained close to their original area (MacArthur et al. 2008). Lobsters have been shown to be more sedentary during the "reds" phase of their life (How et al in prep), though colour alone may not be a good indicator of the degree of lobster movement (MacArthur et al. 2008).

Spillover is often touted as one of the benefits to the fishery of spatial closures (Gell and Roberts 2003). This often takes some time to occur before potential density dependent effects force movement of species out of closures (Abesamis and Russ 2005). While there was no evidence of spillover as part of this study, the continuation of a tagging program as part of a long term monitoring program would potentially provide evidence of spillover. This would be valuable information to demonstrate the effectiveness of marine reserve functioning in terms not only of protection of stock, but supply of post-larval recruits to the fishery.

Information on lobster growth can provide insight into changing environmental conditions and is also important information for stock assessment. As growth is plastic with regard to water temperature (Ehrhardt, 2008) and food availability (Edgar, 1990), continual monitoring will enable changes in the marine environment to be monitored. This is important for the reserves at Rottnest Islands, particularly Parker Point, which is on the southern side of the island. The southern coast of Rottnest Island is effected by the Leeuwin Current which has been shown to influencing fish assemblages (Hutchins 1991 in Hutchins and Pearce 1994). Alteration to oceanographic conditions may alter the influence of the Leeuwin Current on Rottnest Island. Continual monitoring of growth rates of lobsters may provide some insight as the effects of climate change on the demographics of this recreationally and commercially iconic species.

Similarly, size at maturity (SAM) is strongly related to water temperature (Mellville-Smith & de Lestang 2006). The size at maturity recorded as part of this study (79.6 mm CL) is lower than previous estimates from around Perth (Fremantle 87.5 mm CL), which are from deeper water surveys (Mellville-Smith & de Lestang 2006). This may be reflective of the different (warmer) water conditions around Rottnest Islands causing its SAM to be closer to that recorded in more northern latitudes (Jurien Bay 81.4 mm CL and Dongara 74.9 mm CL; Melville-Smith & de Lestang 2006)

3.4.3 Recommendations for a Long-term Monitoring Program

This study provides the basis for a detailed long term monitoring program for lobsters in the Swan Catchment region. It provides baseline data on lobster abundance, size composition, size as well as growth rates and size at maturity.

Changes in size composition and abundance of lobsters within a marine reserve are important gauges as to closure functioning. However to be correctly interpreted they need to be part of a long-term study combined with baseline information. This study has provided baseline levels for three of the four parks being surveys prior to (Shoalwater Islands Marine Park) or immediately after (Rottnest – Parker Point and Armstrong Bay) gazetting. While initial surveys were done after gazetting at Rottnest

Islands, it should be noted that the initial surveys were conducted prior to any extractive pressure being exerted on the open site disproportionately to the closed areas as they were before the commencement of the rock lobster season. Whilst those data haven't been discussed here, they do provide a baseline comparison to any marine reserve survey.

The following are a series of recommendations, and justifications for the successful implementation of a cost effective long-term monitoring program for wester rock lobster throughout the Swan Catch region;

1. Continued annual survey of lobster abundances (ideally conducted during October – November) in each of the four marine reserves

Conducting an annual study will be more cost effective than multiple studies through the year, and provide a standardised sampling period that provides consistency between years.

The recommendation to monitor in October or November has several potential advantages.

- It would allow sampling prior to the commencement of the lobster fishery (15th November) to look at numbers of that year prior to any extraction.
- Remove any biases associated with sampling during the season such as a shift in effort between years.
- Allow the possible use of commercial vessels to pot within in the part enabling more pots to be run, increase the power of any potting analysis, while being cost effective.
- Allow tagging of lobsters prior to any migrations ("whites" migration in November) to look at movement patterns of all lobsters within the marine reserve

2. A multi-disciplinary approach used to assess the dynamics of lobster populations within the four marine reserves.

Potting and UVC provide different yet complimentary techniques to understanding changes in sizes and abundance of lobsters in marine reserves. While demonstrating differing results, the two techniques captured different size distributions within the marine reserves. The multi-disciplinary approach will enable a more holistic examination of the size distribution dynamics and more robust understanding as to potential changes in abundance.

3. Use of different pot types in the potting program

This study has demonstrated a different in the size composition of the lobsters captured by different pot types. As there was no difference in the size composition of commercial and recreational pots, one of these pot types should be used in conjunction with meshed pots to more representatively sample the size distribution of lobsters with the marine reserves.

4. Increased replication of potting and UVC activities.

The differences in potting CPUE between open and closed zones for all parks didn't translate in to significant paired differences at a particular site. This was due the increased statistical power when all sites were combined, compared to the reduced power of a particular within reserve site comparison. Increased potting activity and UVC transects would greatly increase the power of any statistical analysis and allow more certainty to be given to any comparison of paired open and closed areas.

5. Any long-term monitoring study to include a tagging program

As mentioned previously, tagging of animals enables valuable growth information to be obtained, but also vital movement data. To date recaptures of tagging individuals have only had less than 6 months at liberty. While important, this doesn't address issues of long-term site fidelity or ontogenetic changes in movement patterns. Continual tagging and recapture programs will increase the likelihood of recaptures with greater time at liberty, addressing the issue of long-term movement patterns, particularly in closed areas where research potting is the only means of recapture.

4. Fish Communities

P.D. Lewis and M.C. Mackie

4.1. Introduction

Monitoring fish populations and associated communities is crucial for assessing the impacts of fisheries and environmental change. Fishery-independent monitoring methods are particularly important in Perth metropolitan waters where recent bans on commercial fishing have limited the capacity to collect biological samples. The video-based monitoring study described here is therefore important because it aims to:

1. Establish a robust, long-term monitoring program to assess the abundance levels of key fish species within the Perth metropolitan region
2. Describe and contrast fish communities present inside sanctuary zones and in adjacent areas open to fishing within the Perth metropolitan region.

4.2. Methods

4.2.1. Study Areas

Surveys of fish communities were conducted inside sanctuary zones, i.e. areas closed to fishing and in similar nearby areas open to fishing in the following Metropolitan Marine parks.

4.2.1.1. Marmion Marine Park

The Lumps sanctuary zone is relatively shallow (5-8 m) with a number of limestone features that are at or just below the surface. The three sampling sites were spread through the sanctuary adjacent to these structures (Figure 1.1). The open zone sampling sites chosen consisted of two similar limestone features to the south (Wreck Rock and Whitford Rock) and a similar reef area to the north.

4.2.1.2. Shoalwater Islands Marine Park

The closed sites were spread along the seaward/western boundary of the proposed Seal Island Sanctuary, in 5-8 m depth, as most of the sanctuary to the east comprises relatively shallow sand banks. The open sites were spread along the seawardside of Penguin Island in similar depths to those in the sanctuary. Much of the area is relatively shallow and many parts break in moderate swells, as encountered during the spring survey. An additional three sites outside of the sanctuary, to the west and south (depths 6-9m), were sampled in the spring survey to investigate the extent of fish communities encountered.

4.2.1.3. Rottnest Island Marine Reserve

Armstrong Bay sanctuary zone (AB) is located on the north side of Rottnest Island (RI) and as such is not exposed to the full force of the winter swells but does get the effects of larger swell events that wrap around the island to break in areas of the Kingston Reef sanctuary (KR). The sites in AB were spread through the length of the sanctuary at features towards the outer edge of the sanctuary in depths of 11-14m. The open sites were chosen to the east and west of AB at distinct dropoffs into deeper water of 10-15 m.

The KR sites were spread through the established sanctuary zone and selected from the range of sites used by CSIRO in past studies. Each was located adjacent to an area of high relief in depths of 6-8 m.

Parker Point (PP) and Green Island Sanctuary zones (GI) are located on the south side of Rottnest Island and are exposed to the full force of the winter swells and summer seabreezes. The PP sites were spread along the southern edge of the sanctuary in deeper water areas (10-15 m). During the spring survey, the positioning was dictated to some degree by the breaking waves in this area of the sanctuary and much of the sanctuary was inaccessible by boat. The GI sites were again spread along the southern edge of the sanctuary where the reef drops to 15-18 m. The AB open sites were positioned along the bottom of a ridge that drops from 12-18 m off Salmon Bay.

4.2.2. Sampling Design

The surveys were conducted over 9 days in September 2007 (spring survey) and 8 days in January-February 2008 (summer survey). Site selection was primarily aimed at initial description of fish communities with the long-term assessment of changes due to implementation of new sanctuary zones. KR, established as a sanctuary zone since 1988, was included for comparison with its relatively 'pristine' fish community.

Three permanent survey Sites were selected within each Zone for similarity in water depth and habitat (relatively flat locations adjacent to high relief rocky reef). These sites were at least 200 m apart, in the smaller sanctuary zones, and 500 m apart, as suggested by Harvey *et al* (2007), in the larger zones to minimise sampling overlap due to bait plume dispersal. The potential for sampling overlap was assessed using a current drogue deployed a number of times each day in the vicinity of the sampling sites (Section 4.2.4). The sites were surveyed on two consecutive days in each season (surveys at SWIMP during September were a week apart due to weather).

4.2.3. Equipment

Surveys were conducted using stereo baited remote underwater video (SBRUV). Each SBRUV unit consisted of a frame with two Canon HV20 high definition video cameras, each in a waterproof housing mounted so as to provide overlapping camera fields of view at the bait bag, positioned 1.5 m from the cameras (Figure 4.1). A waterproof Sync light (supplied by HandsTek), consisting of a series of LED lights, located within the field of view enabled synchronisation of footage from both cameras so length measurements could be obtained. Each unit was calibrated in a swimming pool with a calibration cube and the CAL software (SeaGIS Pty Ltd) before beginning each survey. The use of the high definition video (HDV) in progressive scan mode (PF25), made measurements easier as the interlacing effect in standard PAL digital video (576i) reduced edge definition, particularly in the faster moving individuals.

Figure 4.1: Picture of stereo BRUV unit used in the study.

4.2.4. SBRUV Deployment

SBRUV units were deployed in groups of three using a 7.5 m vessel. Each SBRUV unit was baited with approximately 500 grams of pilchards and deployed on the seabed such that 60 minutes of HDV footage was obtained. On each sampling day deployment commenced at approximately 08:00 hours to ensure lighting was sufficient for observations and to avoid peaks in dawn feeding behaviour that might bias data. The three SBRUV units were deployed in succession, one per open or closed zone within each area recording the time, depth, SBRUV unit deployed and exact location of each. At the end of each hour-long set the units were retrieved, rebaited, and redeployed in similar fashion at unsurveyed sites. This was repeated to complete the surveys of the three sites within each zone, with retrieval of the last SBRUV occurring around midday to early afternoon, depending on the Area. A current drogue (Figure 4.2.) was also deployed during each survey in close proximity to the study area and allowed to drift for approximately twenty minutes to obtain information on current speed and direction.

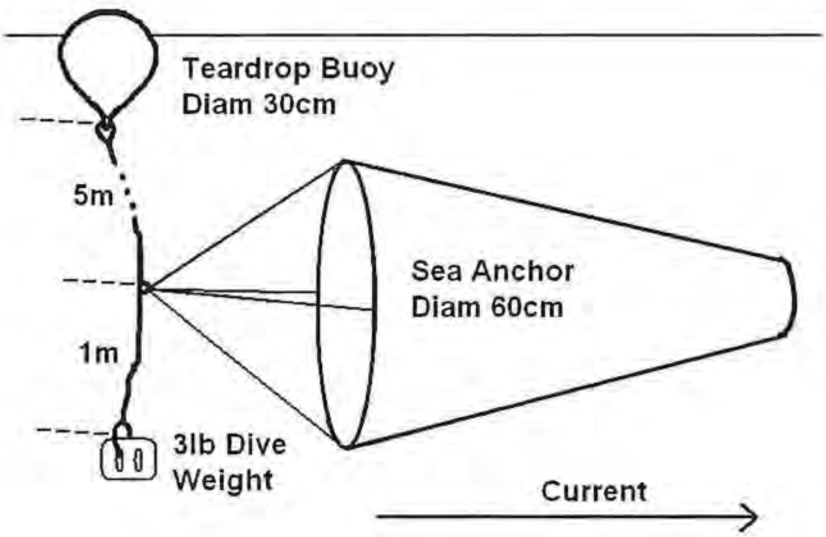

Figure 4.2: Configuration of drogue used to determine current speed and direction.

4.2.5. Data Analysis

Video footage from the left camera of each SBRUV was processed initially using the BRUVS 2.1 database (Cappo and Ericson *pers. comm.*, Australian Institute of Marine Science) to obtain the following parameters:

- Time of First Appearance of each species.
- MaxN (maximum number of individuals in the field of view during the 60 minutes of footage). MaxN provides a conservative estimate of relative

abundance (Willis *et al.* 2000). For large-sized and/or densely schooling species MaxN is likely to be conservative (Mackie et al. 2008).

- Time at which MaxN occurred for all species.
- Activity of each species (feeding, passing, scavenging).
- Time of first feed by each species.
- Habitat characteristics (On reef, on sand near reef, on sand etc).

Once the survey was processed the database was queried for:

- Species richness (number of species at each site).
- Species abundance (MaxN data for each species)
- The time of MaxN for the focal species (see below) to determine the sections of footage to be captured for length analysis (see below).

Other parameters recorded during the processing of tapes were combined into a measure of camera 'effectiveness'. This was a relative measure of the sampling efficiency of each video survey, based on the premise that an ideal survey occurred when the SBRUV unit was deployed on a horizontal plane with a clear view of the reef habitat chosen for survey. To assess the impact of camera 'effectiveness' on number of species and total relative abundance recorded, each of the following parameters were categorised:

- Camera aspect (direction in which the SBRUV unit was facing relative to the reef), with 1 = facing directly at or over the reef, 2 = facing partially towards reef, and 3 = facing directly away from the reef (e.g. over sand).
- Visibility (1 = can clearly see > 2 m beyond bait bag, 2 = can clearly see up to 2 m beyond bait bag, 3 = poor visibility and difficult to see beyond bait bag).
- Surge (linked to visibility; 1 = low surge, 2 = moderate surge where fish can generally maintain position at bait, 3 = high surge where fish cannot maintain position at bait).
- Field of View (1 = clear FOV, 2 = FOV < 50 % obstructed by weed or rock, 3 = FOV > 50 % obstructed).

Following the initial assessment of video footage, length data for focal species were obtained using Photomeasure computer software developed by SeaGIS Pty Ltd. Sections of each pair of tapes for each replicate containing the times of MaxN for the focal species were digitally captured, as audio video interleaved (AVI) files. Playing back in standard DV mode retained the progressive scan properties of high definition recording but reduced the file size of captured footage and need for expensive high definition capture software.

Focal species (n = 12) included those identified by the DoFWA as highly vulnerable to fishing activities (Category 1 species, DoFWA) as well as three abundant species not targeted but occasionally kept by fishers that were abundant at each survey Area (see Section 4.3.1.1). Measurements were of caudal fork length where possible or total length for species without a definite forked tail. The mean distance from cameras at which length measurements were made only varied from 1.5 – 1.8 m between Areas while the maximums were 5.9 m for RI in summer, when the visibility was highest, and 4.0 m for SWIMP and MMP.

Data were subsequently analysed according to the two objectives of the study:

Development of a long-term monitoring program. Data obtained during the initial assessment of video footage were used to determine the best sampling method for future surveys. Patterns in the data for time of First Appearance and MaxN were used to assess the time needed to adequately sample fish communities at each site. Information on current direction and speed was used to assess the possible coverage of the bait plume and hence the minimum distance between survey sites to ensure their independence. The influence of camera deployment was also assessed by comparing camera effectiveness categories with data for times of First Appearance and MaxN. The need for temporal replication was investigated at the daily level by investigating the percentage of species seen on both days and at the seasonal level by comparing species abundances between surveys. The daily replication was further investigated at RI by assigning each species/ species group to one of 3 "response to SBRUV unit" categories (Appendix 2) and comparing the percentage seen on both days for each. Species were assigned to the categories based on their frequency of feeding and behaviour at the camera.

Description and comparison of community structure. The Primer statistical package

(PRIMER-E 2002) was used to describe and compare the species richness and abundance of fish species (i.e. composition of fish assemblages) at each survey site, via the use of a four-way non-parametric permutational multivariate analysis of variance (PERMANOVA). The four fixed factors were Season (Spring, Summer), with Day (1 or 2) nested in Season, and crossed with Area (MMP, RI, SWIMP), with Zone (Open or Closed) nested in Area. Sampling in each zone consisting of three replicate camera sites. Testing the transformation required for the species abundance (MaxN) data, as recommended by Clarke and Warwick (2001) resulted in it being log (x+1) transformed. Non-metric multidimensional scaling (nMDS) techniques, based on the Bray-Curtis similarity coefficient, were applied. Pair-wise analysis was used to determine which differences between sampling units were significant. Where significant variation between fish assemblages were detected the relative contribution of each species to the observed difference was assessed using similarities and percentages (SIMPER) analysis.

The abundance and length data for focal species was compared between areas, seasons, and zones and tested by ANOVA for significant differences with those found to be significant further tested by Tukey HSD posthoc tests for pairwise significance. The SBRUV length data for focal species was also compared to that obtained by the recreational fishing logbook survey (Smith & Hammond 2008, Smith pers. comm.).

4.3. Results

In all, 127 hours of video were processed with a total of 95 species (or species groups) observed during the video surveys (Appendix 2). Focal species selected for further analyses are shown in Table 4.1. These include every 'Category 1' species, listed by the DoF as highly vulnerable to fishing pressure (DoFWA) that was observed during video surveys. It also included three species/ species groups not considered to be under threat by fishing that were abundant in all Areas and hence suitable for comparison of length and abundance data between Areas. Only one other species (Southern maori wrasse) was abundant at all sites but this was not included in analyses due to time constraints. Analysis of the fish communities that these focal species belong to is provided in Section 4.3.2. Detailed examination of the data for each focal species is subsequently provided in Section 4.3.3. These sections are preceded by a review of the methods used in this study in order to determine the best sampling strategy for ongoing monitoring of fish communities (Section 4.3.1.).

4.3.1. Assessment of Survey Techniques

4.3.1.1. Duration of SBRUV deployment

Time of first appearance for all species at each area followed a similar pattern, with 50 % of fish species observed about ten minutes after commencing each survey (Figure 4.3.). The rate at which additional species appeared slowed considerably in an approximately linear fashion after this time, with 80 % of the total numbers of species having been observed after 30 minutes. The mean time of first appearance for the three selected comparative species (Table 4.1) all occurred in the first 15 minutes.

In contrast, only 65 % of the Category 1 species were seen in the first 30 minutes at RI (Figure 4.4) and the rate at which additional species were observed showed little sign of decreasing by the end of the survey period i.e. 60 minutes. Furthermore, the mean time of first appearance for more than half of these species (Table 4.1) are at or after the 25 minute mark. Those two factors indicate that 60 minutes is a minimum survey time to observe many of these species. The exception was blue groper which had a mean time of first appearance of only five minutes, although numbers recorded were low. This indicates that they only come to the camera if they are nearby when the camera hits the bottom and are not attracted to the bait, see Section 4.3.3.1.4.

Data for time at MaxN indicates a similar trend to time of first appearance, with a rapid increase within the first six minutes when MaxN was observed for about 20% of all species. After this point the rate at which MaxN occurred followed a linear trend, with no asymptote in the data by the end of the survey period (Figure 4.5). The mean times of MaxN for the two abundant comparative species (King wrasse and silver/sand trevally) both occurred in the second half of the tape while brownspot wrasse, which were less abundant with a mean MaxN of less than three for all zones (Section 4.3.3.2.2), occurred earlier at 15 minutes (Table 4.1).

The time of MaxN data for Category 1 species is limited and predominantly consists of that for breaksea cod (see n values in Table 4.1) as for other species often only 1 individual was often observed. Nevertheless this data has a linear trend (Figure 4.4), and the mean MaxN data for each species, where available all occurred at or after 30 minutes, (Table 4.1). Thus analysis of the full 60 minutes is required to detect as many Category 1 individuals as possible.

Table 4.1: Mean times (minutes) of first appearance and MaxN for each focal species at Rottnest Island, with counts and standard errors.

	Time First Appearance			Time MaxN		
	Mean	**n**	**Stand. Error**	**Mean**	**n**	**Stand. Error**
a) Category 1 species						
Blue groper	5.24	5	1.95		0	
Queen snapper	13.56	7	6.05		0	
Breaksea cod	15.43	49	2.13	29.91	25	3.0
WA Dhufish	19.08	6	4.53	29.88	1	
Pink snapper	24.55	15	5.13	39.09	6	7.54
Samson fish	25.3	38	2.53	37.84	3	11.09
Yellowtail kingfish	27.93	3	11.15		0	
Harlequinfish	29.93	6	5.73		0	
Baldchin groper	33.59	6	6.40		0	
b) Comparative species						
King wrasse	2.21	71	0.67	33.63	68	2.40
Brownspot wrasse	9.84	70	1.18	15.68	52	1.80
Silver/Sand trevally	14.56	68	1.77	34.34	49	2.30

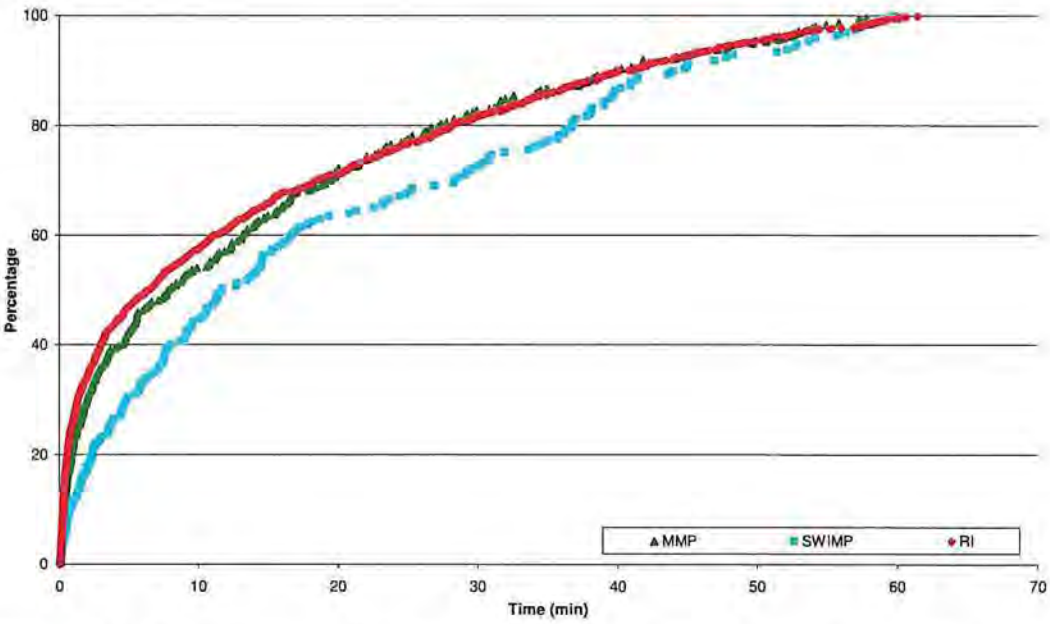

Figure 4.3: Cumulative percentage frequency for time of first appearance of all species observed during video surveys within each area.

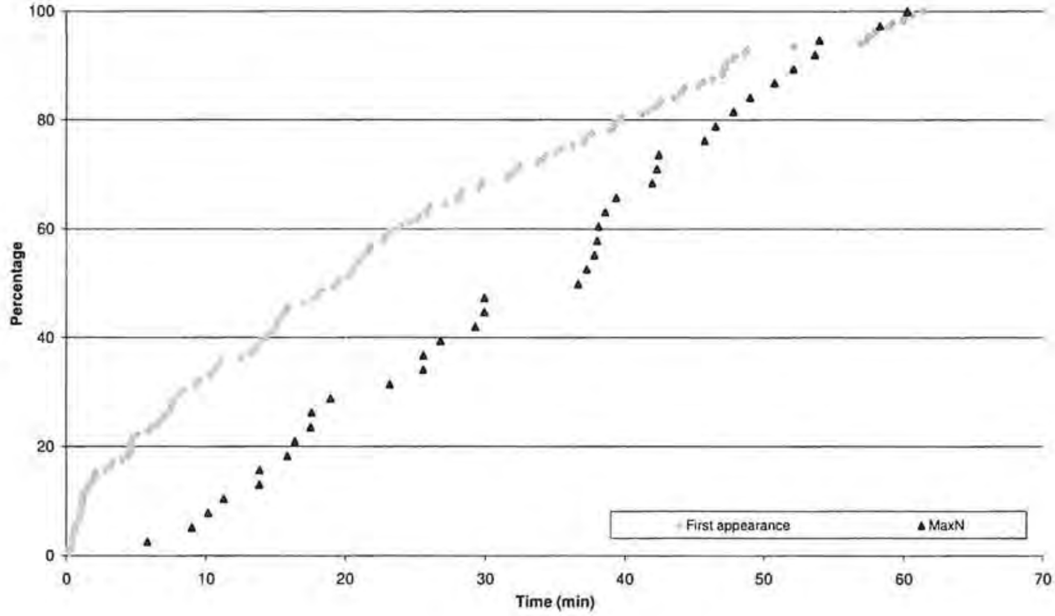

Figure 4.4: Cumulative percentage frequency for time of first appearance and MaxN of Category 1 species observed during video surveys at Rottnest Island.

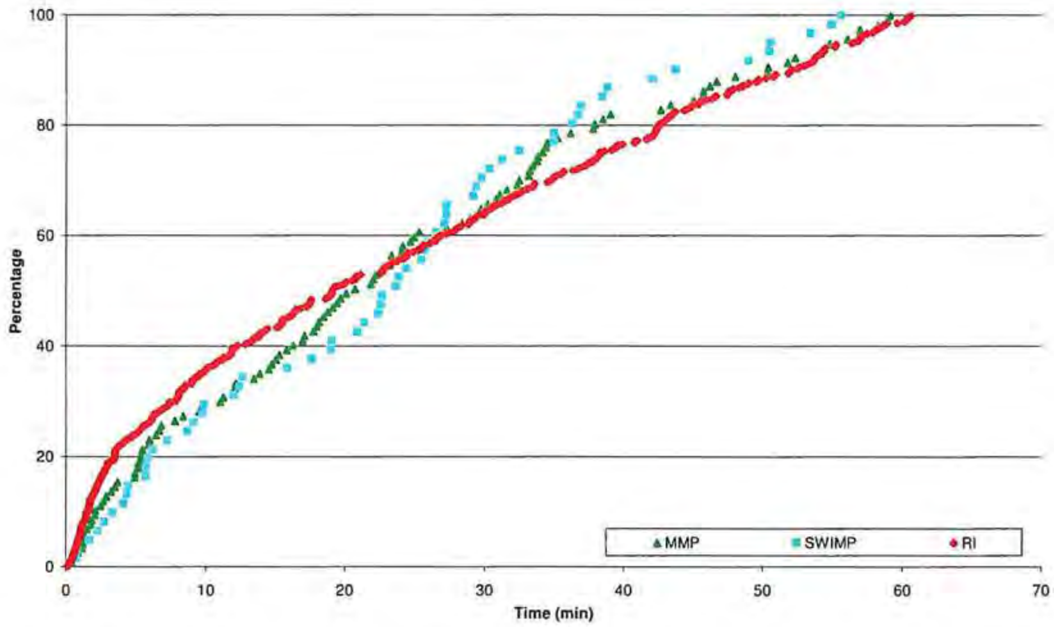

Figure 4.5: Cumulative percentage frequency for time of MaxN of all species observed during video surveys within each area.

4.3.1.2. Daily Replication of Surveys

The number of species observed at each site was generally similar for each of the two days surveyed in each Season (Table 4.2). However, there was considerable difference in the fish species observed between days, with 46 - 67% of species not observed on both days at each Site (Table 4.2). Those species not observed on both days were typically in low overall abundance (less than 10 individuals observed) and included most of the Category 1 species. The relative abundance of species seen on both days also varied widely for example, the MaxN of breaksea cod at AB varied from eleven to two per day on days 1 and 2 of the spring survey, respectively, and from ten to four per day on days 1 and 2 of the summer survey, respectively. Furthermore, at Rottnest Island the maximum number of species/ species groups observed during an individual survey, of 60 minutes, was twenty-seven whereas the total number of species/ species groups observed at RI over both surveys (a total of 36 hours) was eighty two.

These data indicate that many species present at a particular site may not be observed during each survey. This reflects species-specific differences in the response to the SBRUVs, with those species that exhibit a positive response and actively feed at the bait much more likely to be observed on both of the survey days compared to those species that showed little response to the bait and video units (Figure 4.6).

In addition to increased numbers of species, repeat surveys at the same site over consecutive days also decreased the likelihood of a site not being surveyed due to factors such as equipment malfunction.

Table 4.2: Species richness and abundance of species on each day in each area by season.

	Species Richness			Abundance	
	Day1	Day2	Mean % of sp.not recorded on both days	Day1	Day2
a) Spring					
MMP	32	26	50.2	248	214
NRI	41	40	52.5	484	623
SRI	44	43	46.6	499	545
SWIMP	19	24	68.0	73	98
b) Summer					
MMP	35	28	62.3	281	183
NRI	41	40	60.1	484	495
SRI	43	34	54.5	722	686
SWIMP	19	23	60.0	124	182

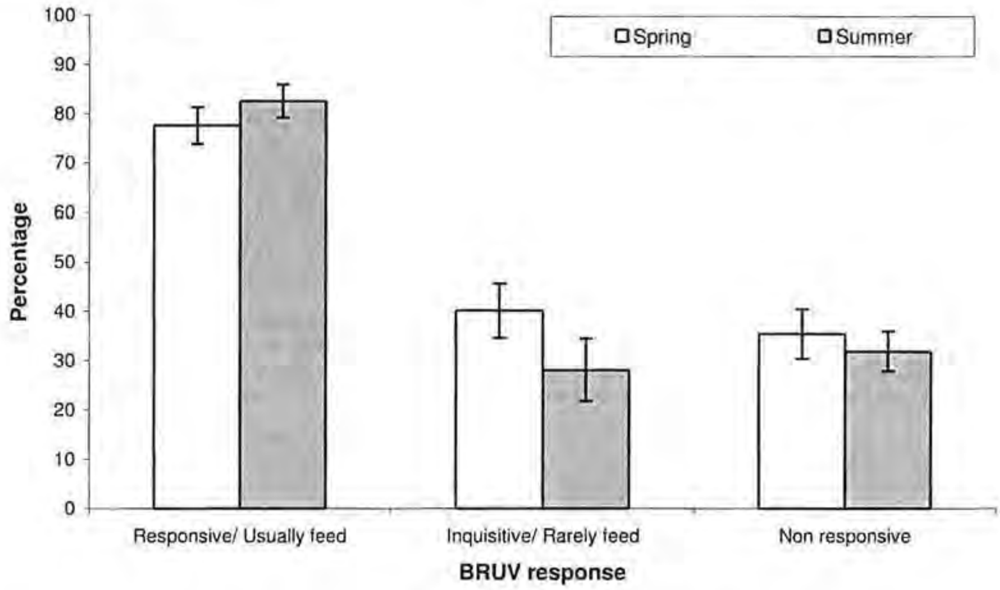

Figure 4.6: Percentage of species seen on both days at Rottnest Island by species response to BRUVs category, with standard errors.

4.3.1.3. Seasonal Replication of Surveys

Although only the GI closed zone at Rottnest and the MMP closed zones had significant differences in fish assemblages between seasons (Section 4.3.2) there were clear differences in the abundance of many species in each Season (see Table 4.3).

For example, the total MaxN of Category 1 species at Rottnest Island was higher during the spring survey (n = 105) than in the summer survey (n = 87). In particular, the total MaxN of dhufish, pink snapper, blue groper and baldchin groper decreased from twenty-seven in spring to eleven in summer. Nevertheless, the total MaxN of breaksea cod remained similar in each Season, with 44 and 45 in spring and summer (respectively), and the total MaxN of Samson fish was higher in summer (n=20) than in spring (n=15).

4.3.1.4. Effects of Environmental Factors and SBRUV Deployment

Investigations into the possible influence of environmental conditions such as visibility, surge, habitat and camera position on the number of species and abundance detected at Rottnest Island sites indicated no obvious relationships. The number of species and abundance of species at Rottnest Island did not differ with current strength (Figure 4.7) and similarly no relationship was observed with Time of Day. Similarly, camera "effectiveness"; (the combined score for camera aspect, water visibility and sea swell) did not appear to affect the number of species and abundance (Figure 4.8). There was some indication that the number of species and abundance were reduced in surveys in which the camera field of view was obstructed (Figure 4.9). However, these reductions were not statistically significant (ANOVA number of species: df = 1, F = 1. 136, P = 0.308; species abundance: df = 1, F = 1.416, P = 0.257). Given these results, all data, including that for surveys in which the camera field of view was obstructed, were used in subsequent analysis of community structure.

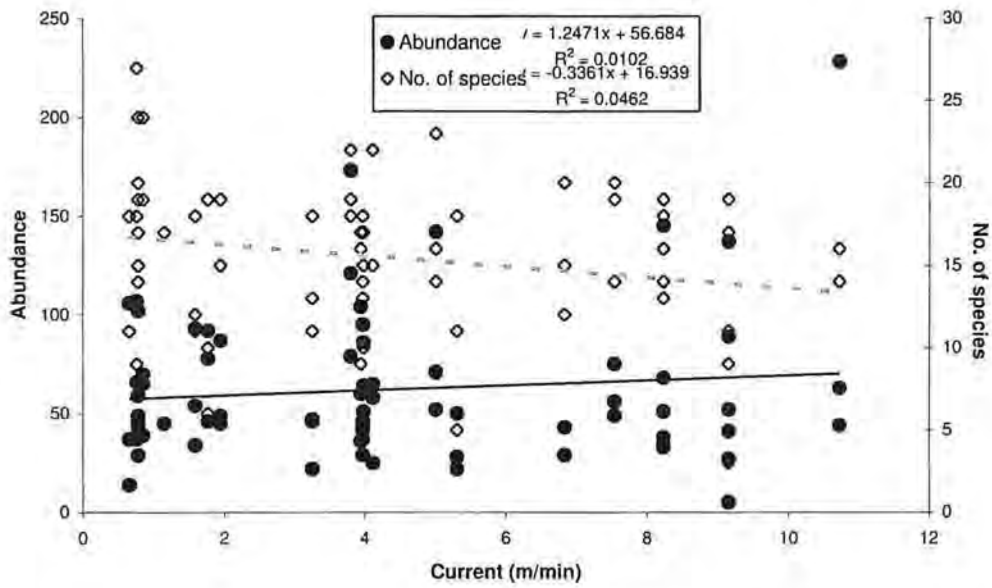

Figure 4.7: Variation in abundance and number of species detected in fish communities at RI with current strength, giving trendlines and regression equations.

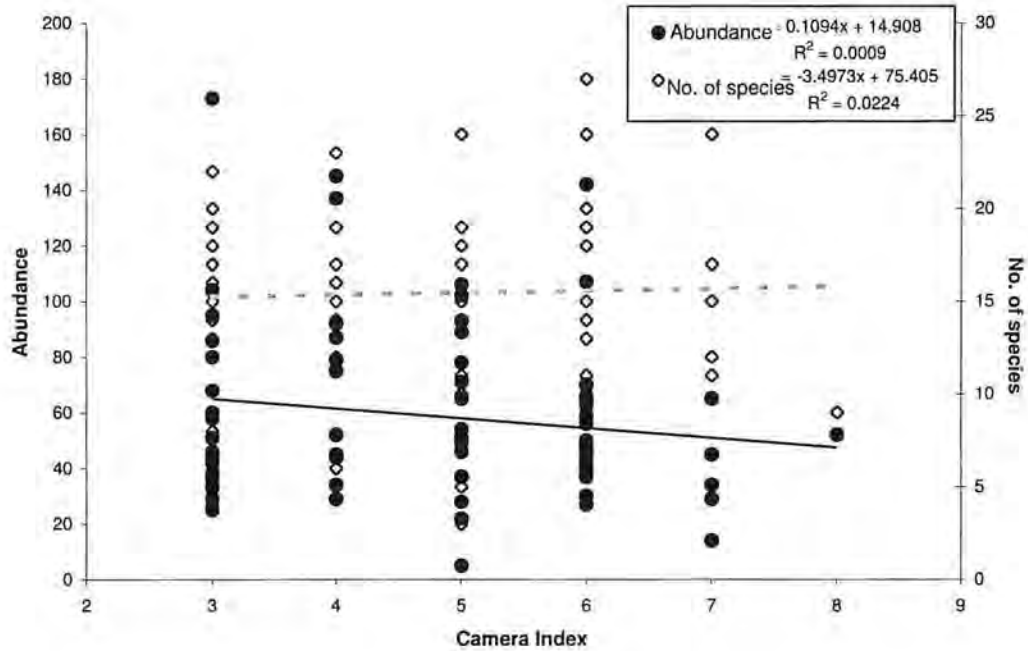

Figure 4.8: Variation in abundance and number of species detected in fish communities at RI with Index of camera effectiveness.

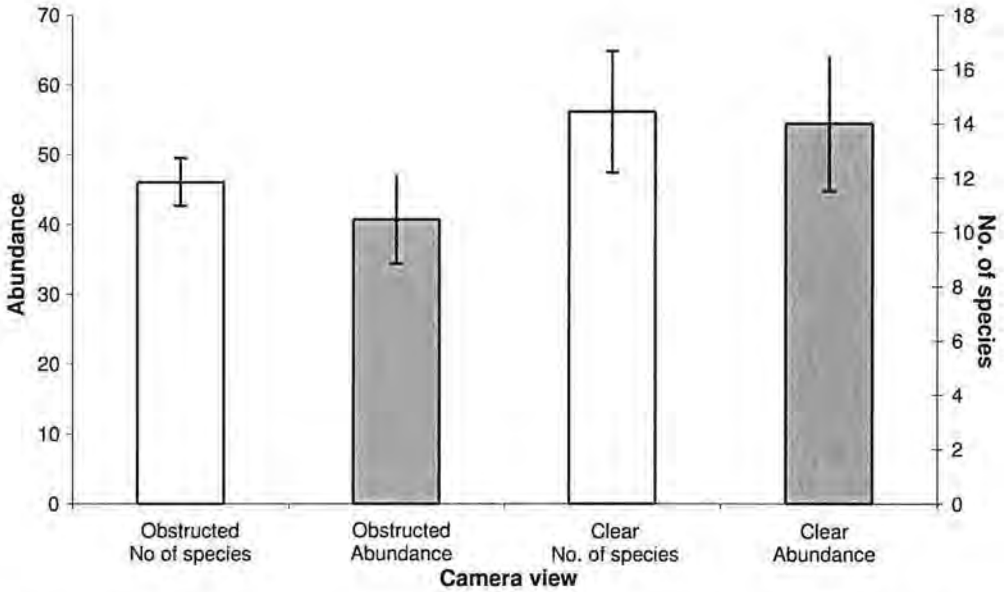

Figure 4.9: Mean number of species and abundance of species, with standard errors, for camera drops with obstructed fields of view and drops at the same Sites in the same Season with clear views.

4.3.1.5. Effect of Current on Fish Assemblage

The size and nature of the surveyed sanctuary zones often restricted the distance between replicate sites to about 200 m, particularly in SWIMP and MMP. In comparison, the currents measured during the surveys varied widely and the bait plume emanating from each SBRUV was estimated to travel between 40- 640 m during the survey period (assuming a direct relation with current strength). The currents measured at SWIMP were low (0.3-3.1 m/min) and even though there was minimal separation of survey sites, due to the small size of the sanctuary, there was no overlap in bait plume with adjacent sites. At MMP, however, the measured currents were higher (0.4-5.4 m/min) resulting in 1 day during summer when the possible distance travelled by the bait plume (324 m) exceeded the distance between two sites and the southerly direction of the current would allow overlap of plume with an adjacent site. At RI there was a wide range in the measured current (0.7-10.7 m/min). Both south and north RI had one day in the summer surveys with high currents where the possible distance travelled by the bait plume (NRI 410-494 m, SRI 550-640 m) exceeded the distance between some survey sites. On both days the current was moving in a west to northwesterly direction that would allow the bait plume to cross some of the adjacent sites. Although for both MMP and RI these adjacent sites were not being surveyed at the time.

4.3.2. Comparison and Description of Fish Assemblages

A total of 95 different species/groups were observed during the surveys. The largest number, 82 species, were observed within the RI Area whereas 49 species were observed in the MMP and 35 in the SWIMP (Table 4.3). The most abundant species were the western king wrasse, footballer sweep, silver/sand trevally and southern maori wrasse, which comprised 24, 12, 11 and 6% of the total number of fish observed during the surveys, respectively. In contrast, there were 40 species that were observed only five or less times during both surveys, highlighting the rarity or low responsiveness to the SBRUVs of many species.

Comparison of the average Shannon H diversity index between seasons and zones within each area (Figure 4.10) showed few seasonal differences, with the exception of PP, but a number of differences between areas. Notably, SWIMP having lower diversity than the other areas.

Table 4.3: Overall total relative abundance (Sum of MaxNs from each replicate) of each fish species/ species group observed in each Zone of each Area by Season, with total numbers seen and total No. of species. Category 1 species shown in bold and comparative species in italics.

	Season	Spring										Summer										
	Area	MMP				RI				SWIMP		MMP				RI				SWIMP		
Common Name	Zone	Op	Clsd	AB	AB Op	KR	GI	PP Op	PP	Op	Clsd	Op	Clsd	AB	AB Op	KR	GI	PP Op	PP	Op	Clsd	Total
Western king wrasse		39	33	145	175	95	65	135	37	2	3	41	30	110	69	165	19	88	84	22	16	1373
Footballer sweep				1	66	1	112	106	14			6	34	100	16		54	83	92			685
Silver/Sand trevally		14	29	43	52	11	18	50	20	3	17	116	1	78	25	27	30	20	20	42	18	634
Southern Maori wrasse		5	9	21	32	18	13	21	32	6	3	4	6	25	30	19	12	14	25	5	2	302
Rough bullseye		10	3				5	1	1			2	10				209	20	40		1	302
Brownspot wrasse		15	13	15	19	16	15	8	15	9	11	13	14	13	14	16	9	6	11	13	12	257
Silver drummer		5	48	7	20	3	4	3	4			3	7	1	6	2	3	1	132		1	250
Red-banded wrasse		15	3	19	17	11	18	8	18			5	5	15	11	8	19	6	12		1	191
McCulloch's scalyfin		25	17	5	16	12	7	4	5			14	9	8	7	9	9	3	8		2	160
Black head puller			2	1	20	2	15	6	21			6					2		82			157
Westralian puller						2	44		6								58	6	17			133
Breaksea cod				13	7	7	7	5	5					14	6	4	5	5	11			89
Striped seapike													3								85	88
Black spot wrasse		3	3	5	7	8	6	3	12	1		4	7	3	4	5	3	3	9			86
Western buffalo bream		32										35			1	7						75
Senator wrasse		6	4	4	4	5	5	1	9	1		5	5	1	3	1	3	3	6	1		67
Scribbled chiseltooth wrasse				4	7	7	6	5	12					3	7	3	7		2			63
Yellowtail scad														2	5	1	44	1	6			59
Horseshoe leatherjacket		3	7	7	7	7	1		3				3	2	6	5			3			54
Banded sweep		5	2	1	5		6	1	4	3	2	1	1	2	2	1		2	5	6	3	52
Silverbelly								3	5	9	7							1	2	6	14	47
Samson fish		4	3	6	1	5	1	2			2			5	3	3	3	4	2		1	45
Blue lined leatherjacket				6	5	4	4		4	1	1		1	5	1	5	3	1	2			43
Woodwards reef eel		2	3				4	1	2	2		2	4	2	2	2	3	2	3	4	3	41
Western talma		7	1	1	3	2	4					5	8	3		3	1		4			42
Herring cale		3	3	2	5	2	4	2		1	1	3	1		2			1	3	1	2	36
Eagle ray		1	1			1	2	2	1	1	2	4	2	1	1	1		2	2	5	1	30
Smooth stingray		5	1	1	1		4	1	1		3	3	1	4	2			1			3	31
Australian herring			1							3	1	2				6				12	3	28
Port Jackson shark		4	2	1		2	1	6	5			1	1				1	1	1	1		27
Yellow Headed hulafish			17										1				7					25
Pink snapper		2		2		5		2	4				1	1		6						23
King George whiting				1	3	2		1	2					3	3	2				1		18
Striped stingaree		1		3	2		1	2	3				3			1		1				17
Western foxfish				2	1	1	1	2	1					1	1	2	2		2			16
Sea sweep		2	1	2	2	1		1						2	2		1		2			16
Striped toadfish												10								5	1	16
Green moon wrasse				1			4									1	4	2	1			13
Blackspot goatfish						1						1	1	1	1	8						13
Blue spotted goatfish					1			2			2					1		1	1		5	13
Lnid Wrasse		1	2				5									3						11
Green Moray				1	2	3	1											1	2			10
Whiting										7	3											10
Black spot catshark						1		1		1	2		1						1			7
Queen snapper					2				2						1		2					7
WA dhufish						2	1	3									1					7

Snakeskin wrasse						1	2	2							1			1			7
Zebrafish			2		1			1					1	1							6
Goldspot sweetlip	1				1						1				2						5
Baldchin groper			2	1	1									2							6
Moon wrasse							1	2					1			1	1				6
Yellow-striped leatherjacket												1				4	1				6
Western wobbegong						1						2				1	2				6
Western wirrah		1			1				2			1			1						6
Harlequin fish			1		1									1	2	1					6
Red striped cardinalfish											3	2									5
Yellowtail kingfish					2	1							1	1							5
Blue groper			1	2	1											1					5
Dusky morwong	1	1											1							1	4
Old wife						2									2						4
Rainbow cale	2	1									1										4
Tarwhine	1		1								1									1	4
Silver spot						1	1	1													3
Brownfields wrasse												1								2	3
Western butterfish										1										2	3
White-barred boxfish													1		2						3
Southern blue-spot flathead																	2		1		3
Western sea carp									1						1						2
Swallowtail																2					2
Fusilier sweep													2								2
Redlipped morwong															2						2
Western blue devil						1											1				2
Fiddler ray							1	1													2
Black banded seaperch			1			1															2
Snook					2																2
Toadfish																	2				2
Rainbow runner								1													1
Shark sp				1																	1
Crested morwong	1																				1
Slender suckerfish				1																	1
Cleaner fish																1					1
False senator wrasse												1									1
Spiny-tailed leatherjacket																1					1
Beardie						1															1
Wavy grubfish												1									1
Goatfish																				1	1
Sharpnosed weed whiting	1																				1
Western smooth boxfish													1								1
Bicolor scalyfin											1										1
Victorian scalyfin											1										1
Lined dottyback						1															1
Spotted gurnard fish												1									1
Whiskery shark								1													1
Unid. Gurnard										1											1
Individuals	**216**	**211**	**328**	**487**	**247**	**394**	**393**	**257**	**53**	**62**	**294**	**170**	**413**	**236**	**330**	**526**	**288**	**594**	**125**	**181**	**5805**
No. Species	**30**	**27**	**34**	**31**	**37**	**40**	**34**	**35**	**17**	**17**	**29**	**34**	**33**	**31**	**38**	**35**	**33**	**33**	**15**	**24**	

Multivariate analyses indicated significant differences in fish assemblages between Areas, Seasons, Days, Zones (Table 4.4.) and importantly between each Area in each Season (Table 4.5.). However, there were no differences in fish assemblages observed in each Area or Zone during surveys on replicate days (Table 4.4.). Nevertheless, as detailed in Section 4.3.1.2. there were distinct differences in the number of species observed at each Site on the replicate days due to species-specific responses to the SBRUV units.

Fish assemblages within the SWIMP were particularly distinct and exhibited much more intra-Area variability than did fish assemblages within the MMP and RI (Figure 4.11.). SIMPER analysis of species abundance within each Area indicate that differences in fish assemblages are mainly due to differences in abundances of the more common species, with ten species accounting for 46 – 52% of observed differences between Areas (Table 4.5.). Five of the most abundant species in particular were among these ten in each of the Area comparisons, including king wrasse, trevally, red-banded wrasse, southern maori wrasse and McCulloch's scalyfin. All of these had a ratio of dissimilarity to standard deviation greater than 0.75, indicating they contributed significantly to the observed differences. Only one Category 1 species, breaksea cod, contributed significantly to the observed differences in fish assemblages, between RI and the other two Areas. With the exception of silverbelly, all of the ten species that contributed most to the differences were in lower abundances or absent in the SWIMP compared to the MMP and RI.

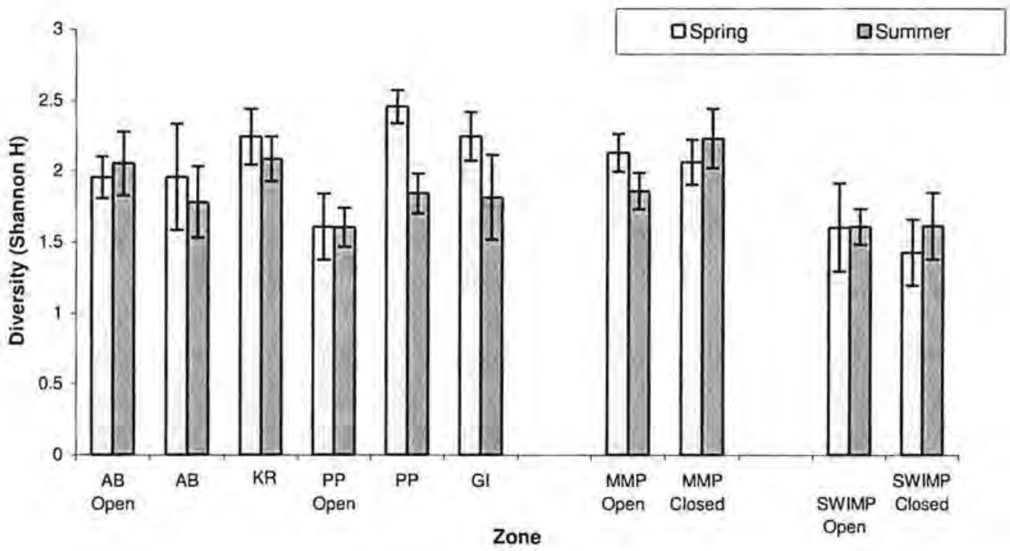

Figure 4.10: Mean Shannon H diversity index for fish assemblages in each Zone by Season, with standard errors.

Table 4.4: Results from PERMANOVA tests of ln(x+1) transformed fish abundance data for differences between factors in both surveys. Bold values indicate a significant difference at $p < 0.05$.

Source	df	MS	pseudo-*F*	p(perm)
Season	1	3219.7	2.2876	**0.0034**
Area	2	25204	17.908	**0.0002**
Day (Season)	2	2152.4	1.5293	**0.0418**
Zone (Area)	7	3782	2.6872	**0.0002**
Season X Area	2	2792.4	1.984	**0.0012**
Season X Zone (Area)	7	1305.8	0.92778	0.6682
Day(Season) X Area	4	1760.4	1.2508	0.1112
Day(Season) X Zone (Area)	14	856.4	0.60848	1
Residual	78	1407.4		
Total	117			

Table 4.5: Pairwise analysis of differences in fish assemblages between Areas for both surveys and by each Season. Bold values indicate significance at $p<0.05$.

	Both Surveys		**Spring**		**Summer**	
Comparison	**t**	**p**(perm)	**t**	**p**(perm)	**t**	**p**(perm)
MMP, SWIMP	3.3042	**0.0002**	2.5198	**0.0002**	2.5656	**0.0002**
MMP, RI	3.5688	**0.0004**	2.6409	**0.0002**	2.8157	**0.0002**
RI, SWIMP	5.1709	**0.0002**	3.6655	**0.0002**	3.9063	**0.0002**

Table 4.6: Results of SIMPER analysis for Areas, giving the top 10 species and the positions of the category 1 species. Species that occur in all comparisons are shown in bold. Bold values indicate significant ratio of dissimilarity to standard deviation > 0.75.

Species	Av. Abund	Av. Abund	Av. Diss	Diss /SD	Contrib %	Cum. %
	MMP	**RI**				
Sand/Silver trevally	1.21	1.49	4.21	**1.39**	7.24	7.24
Footballer sweep	0.32	1.26	4.02	**0.98**	6.91	14.15
King wrasse	1.81	2.49	3.58	**1.25**	6.15	20.30
Maori wrasse	0.62	1.42	2.77	**1.68**	4.77	25.07
Red-banded wrasse	0.60	1.06	2.41	**1.30**	4.14	29.22
Silver drummer	0.60	0.48	2.35	**0.79**	4.05	33.27
McCullochs scalyfin	1.21	0.74	2.10	**1.07**	3.62	36.88
Breaksea cod	0.00	0.68	2.08	**1.25**	3.57	40.46
Rough bullseye	0.39	0.27	1.85	0.62	3.19	43.64
Western talma	0.53	0.20	1.65	**1.11**	2.84	46.48
17. Samson fish	0.17	0.31	1.25	**0.91**	2.14	63.03
27. Pink snapper	0.07	0.16	0.69	0.52	1.19	80.14
30. King George whiting	0.00	0.16	0.54	0.53	0.94	83.22
	RI	**SWIMP**				
King wrasse	2.49	0.61	8.71	**1.43**	11.76	11.76
Footballer sweep	1.26	0.00	4.75	**0.89**	6.41	18.17
Sand/Silver trevally	1.49	0.98	4.55	**1.13**	6.15	24.32
Maori wrasse	1.42	0.54	4.12	**1.46**	5.56	29.89
Red-banded wrasse	1.06	0.05	4.11	**1.79**	5.55	35.44
McCullochs scalyfin	0.74	0.07	2.78	**1.51**	3.75	39.19
Breaksea cod	0.68	0.00	2.64	**1.22**	3.57	42.76
Black-spot wrasse	0.56	0.05	2.19	**1.16**	2.96	45.72
Silverbelly	0.09	0.47	2.07	0.66	2.79	48.51
Brownspot wrasse	1.09	0.99	2.03	**0.91**	2.74	51.25
19. Samson fish	0.31	0.08	1.37	**0.77**	1.86	69.88
28. King George whiting	0.16	0.03	0.75	0.56	1.01	82.93
31. Pink snapper	0.16	0.00	0.70	0.44	0.94	85.81
	MMP	**SWIMP**				
King wrasse	1.81	0.61	7.10	**1.71**	9.89	9.89

McCullochs scalyfin	1.21	0.07	5.95	**2.26**	8.29	18.18
Sand/Silver trevally	1.21	0.98	5.88	**1.33**	8.19	26.37
Silver drummer	0.60	0.05	3.16	0.70	4.40	30.77
Red-banded wrasse	0.60	0.05	2.88	**1.03**	4.01	34.78
Maori wrasse	0.62	0.54	2.73	**1.21**	3.80	38.58
Senator wrasse	0.57	0.13	2.47	**1.39**	3.43	42.02
Western talma	0.53	0.00	2.43	**1.09**	3.39	45.41
Silverbelly	0.00	0.47	2.26	0.61	3.15	48.56
Black-spot wrasse	0.47	0.05	2.18	**1.14**	3.04	51.60
24. Samson fish	0.17	0.08	1.11	0.60	1.54	83.90

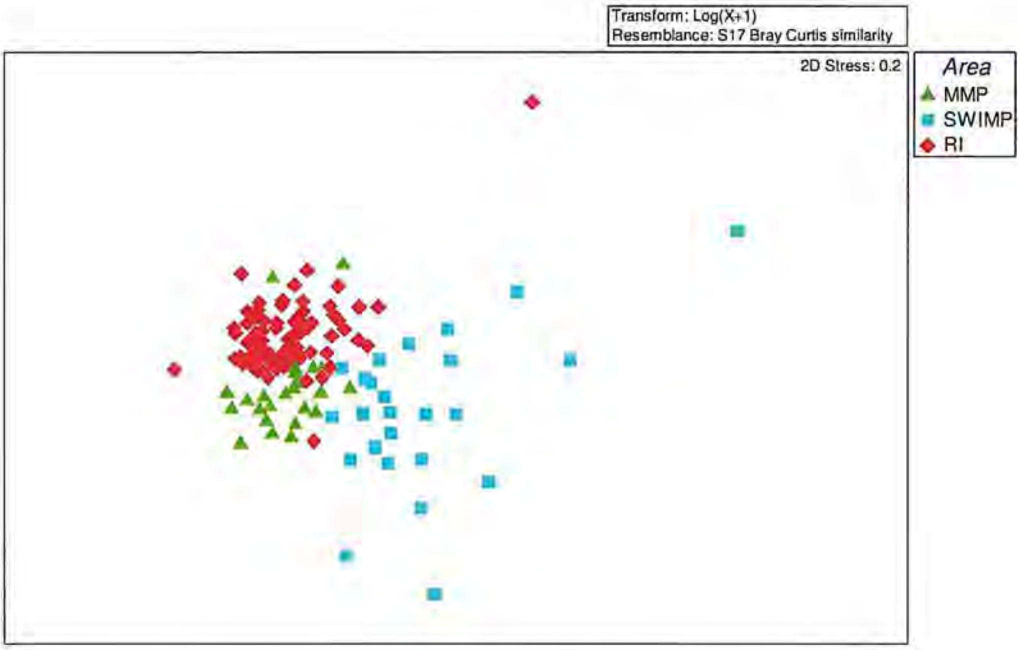

Figure 4.11: Non-metric MDS ordination of fish assemblages observed during SBRUV surveys of all Sites within each Area.

4.3.2.1. Description of Fish Assemblages at Rottnest Island

Fish assemblages observed during replicate surveys within each Zone at RI were similar (Figure 4.12) being comprised of mainly temperate species (70–85 %) with some subtropical (15-27 %) and a few tropical species (1-9 %). The similarities in fish assemblages between Zones, particularly those adjacent to each other is evident in Figure 4.12. For example, fish assemblages within Zones on the south side of RI

(GI, PP and PP Open) were more similar to each other than they were to those on the north side (KR, AB and AB Open). Many species contribute to this north-south difference including lower abundances of Footballer sweep, pullers and banded sweep plus higher abundances of king wrasse, trevally, and breaksea cod at NRI Zones compared to SRI (Table 4.7.).

Fish assemblages within KR were generally distinct from those in all other Zones (Table 4.8.). During the summer surveys they differed from all other Zones except AB, whilst in spring they were only similar to those assemblages observed within AB Open (Table 4.8.). The strongest and most consistent similarities in fish assemblages occurred between AB and AB Open. Differences in the abundance of many species contribute to these significant differences in fish assemblages between Zones but it is primarily due to variations in the abundance of common species (eg Table 4.9 gives comparison of KR and GI, the Zones most distinctly separated in nMDS). The absence of three species (Rough bullseye, pink snapper and blackspot goatfish) and low abundances of other species, such as footballer sweep, horseshoe leatherjacket, Westralian puller and black headed puller, in one Zone or the other contributed to the observed difference in fish assemblage between these 2 most distinctly different RI Zones.

Even though there was a significant Seasonal difference between the fish assemblages of RI ($t=1.5213$, $P= 0.005$) and seasonal differences in Category 1 species abundances were evident (Table 4.3, Section 4.3.1.3) when investigated by Zone only GI had significantly different assemblages between Seasons ($t= 1.4203$, $P=0.024$). The absence of four species (Banded sweep, yellowtail scad, smooth stingray and herring cale) from one Season plus variations in the abundance of other species contributed to this difference (Table 4.10).

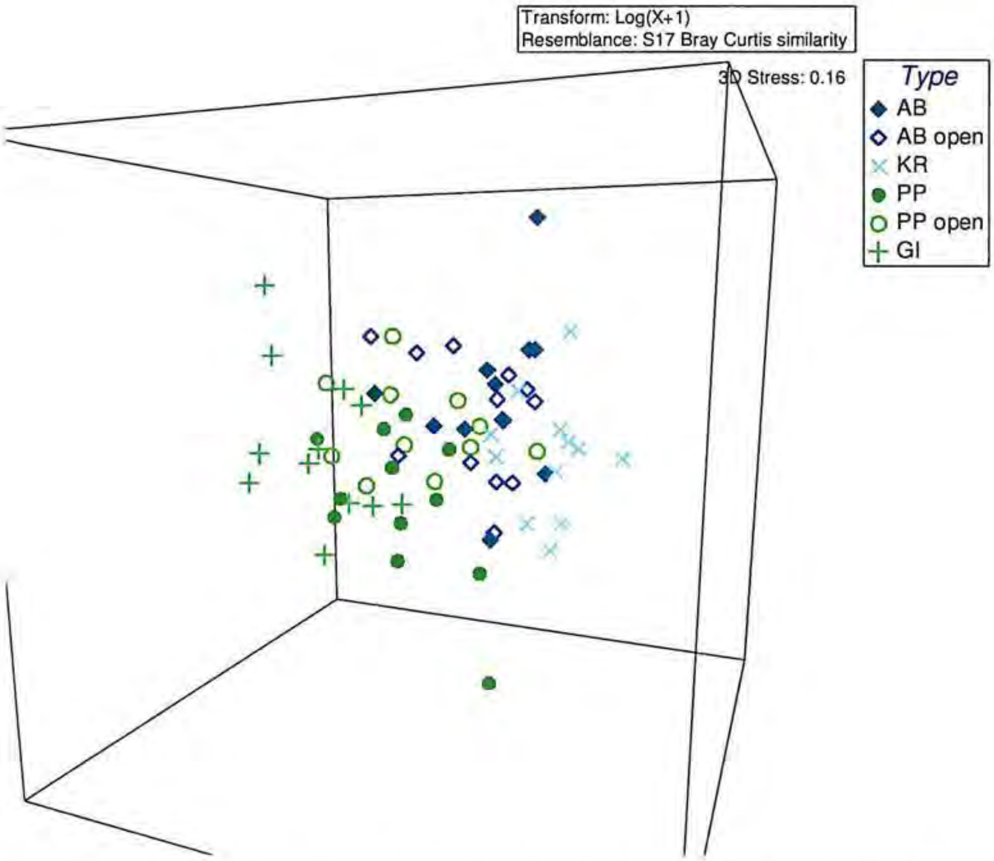

Figure 4.12: Non-metric MDS ordination of fish assemblages observed during SBRUV surveys of Sites in each Zone at Rottnest Island.

Table 4.7: Results of SIMPER analysis for differences in fish assemblages between NRI and SRI Zones. Category 1 species are shown in bold. Species in italics indicate absence from one Zone. Bold values indicate significant ratio of dissimilarity to standard deviation > 0.75.

Species	NRI Av. Abund	SRI Av. Abund	Av. Diss	Diss /SD	Contrib %	Cum. %
Footballer sweep	0.44	2.04	5.54	1.69	10.36	10.36
King wrasse	2.72	2.21	3.56	1.00	6.66	17.02
Sand/Silver trevally	1.55	1.40	2.90	1.11	5.42	22.44
Westralian puller	0.03	0.64	1.82	0.60	3.40	25.84
Redbanded wrasse	1.06	1.04	1.82	1.08	3.40	29.24
Breaksea cod	0.74	0.59	1.82	1.14	3.39	32.63
Silver drummer	0.45	0.50	1.79	0.72	3.34	35.96
Maori wrasse	1.51	1.35	1.63	0.88	3.05	39.02
McCullochs scalyfin	0.81	0.62	1.56	1.06	2.91	41.92
Black head puller	0.17	0.54	1.56	0.61	2.91	44.83
Horseshoe leatherjacket	0.54	0.12	1.51	1.05	2.81	47.65
Scribbled chieseltooth wrasse	0.43	0.33	1.49	0.83	2.79	50.44
Rough bullseye	*0.00*	0.53	1.49	0.44	2.78	53.22
Black spot wrasse	0.55	0.58	1.48	1.13	2.76	55.98
Brownspot wrasse	1.22	0.96	1.41	0.98	2.64	58.63
Samson fish	0.40	0.23	1.18	0.91	2.20	60.82
Senator wrasse	0.34	0.48	1.17	0.98	2.19	63.01
Blue lined leatherjacket	0.48	0.27	1.16	1.01	2.16	65.17
Banded sweep	0.21	0.31	1.01	0.87	1.89	67.06
King George whiting	0.27	0.06	0.90	0.79	1.69	68.74
Woodwards reef eel	0.12	0.26	0.84	0.71	1.56	70.30
Western talma	0.22	0.16	0.83	0.77	1.54	71.85
Pink snapper	0.22	0.11	0.81	0.63	1.51	73.35
Port Jackson shark	0.06	0.26	0.80	0.66	1.50	74.85
Herring cale	0.20	0.19	0.76	0.77	1.42	76.27

Table 4.8: Pairwise PERANOVA comparison of fish assemblages between RI Zones for both Seasons and within each Season. Bold values indicate significance at p<0.05.

	Combined		Spring		Summer	
Comparison	**t**	**p** (perm)	**t**	**p** (perm)	**t**	**p** (perm)
KR, AB open	1.5689	**0.0104**	1.3678	0.074	1.3593	0.0558
KR, AB	1.612	**0.0088**	0.9801	0.4436	1.5468	**0.0166**
KR, PP	2.0533	**0.0004**	1.4378	**0.03**	1.9907	**0.0024**
KR, PP open	2.3885	**0.0004**	1.7198	**0.0284**	2.0836	**0.0044**
KR, GI	2.8215	**0.0002**	1.8909	**0.001**	2.5204	**0.0016**
AB open, AB	1.171	0.216	1.2387	0.1842	0.9768	0.508
AB open, PP	1.5674	**0.0142**	1.4065	0.0798	1.4603	**0.0458**
AB open, PP open	1.9592	**0.003**	1.4478	0.1124	1.579	**0.02**
AB open, GI	2.3106	**0.0004**	1.4598	**0.0384**	1.9322	**0.0098**
AB, PP	1.9544	**0.0004**	1.3499	0.0798	1.768	**0.0066**
AB, PP open	2.1594	**0.0016**	1.7056	**0.0318**	1.6709	**0.0268**
AB, GI	2.5052	**0.0004**	1.9403	**0.0028**	2.0576	**0.0032**
PP, PP open	1.7894	**0.0082**	1.5032	0.067	1.2282	0.1998
PP, GI	1.5489	**0.009**	1.5219	**0.0262**	1.4007	**0.0342**
PP open, GI	1.8998	**0.011**	1.7218	**0.0486**	1.1851	0.2114

Table 4.9: Results of SIMPER analysis between KR and GI Zones at RI. Category 1 species are shown in bold. Species in italics indicate absence from one Zone. Bold values indicate significant ratio of dissimilarity to standard deviation > 0.75.

Species	**KR** Av. Abund	**GI** Av. Abund	Av. Diss	Diss /SD	Contrib %	Cum. %
Footballer sweep	0.06	2.45	6.64	**2.81**	11.54	11.54
King wrasse	2.90	1.85	3.84	**1.44**	6.67	18.21
Westralian puller	0.09	1.24	3.33	**0.89**	5.79	24.00
Sand/Silver trevally	0.96	1.42	2.62	**1.47**	4.55	28.54
Rough bullseye	*0.00*	0.91	2.58	0.61	4.49	33.03
Scribbled chieseltooth wrasse	0.50	0.42	1.55	**1.12**	2.69	35.72
Redbanded wrasse	0.90	1.33	1.53	**1.42**	2.67	38.39
Horseshoe leatherjacket	0.61	0.06	1.50	**1.36**	2.61	40.99
Black head puller	0.09	0.53	1.49	0.71	2.59	43.58
Maori wrasse	1.34	1.05	1.44	**0.93**	2.50	46.08
McCullochs scalyfin	0.91	0.78	1.41	**1.36**	2.45	48.53
Black spot wrasse	0.65	0.45	1.39	**1.15**	2.42	50.95
Pink snapper	0.50	*0.00*	1.35	**0.92**	2.35	53.30
Breaksea cod	0.54	0.62	1.29	**1.15**	2.24	55.54
Blackspot goatfish	0.44	*0.00*	1.21	**0.89**	2.10	57.65
Brownspot wrasse	1.27	1.04	1.17	**1.33**	2.04	59.69
Samson fish	0.46	0.23	1.14	**1.19**	1.99	61.67
Green moon wrasse	0.06	0.40	1.12	**0.95**	1.95	63.62
Western talma	0.29	0.26	1.04	**0.99**	1.81	65.43
Woodwards reef eel	0.12	0.38	1.02	**0.92**	1.77	67.20
Blue lined leatherjacket	0.50	0.40	1.01	**0.95**	1.76	68.96
Banded sweep	0.06	0.32	1.00	**0.95**	1.74	70.70
Silver drummer	0.29	0.40	0.95	**0.96**	1.66	72.36

Table 4.10: Results of SIMPER analysis of GI Zone at RI between Seasons. Category 1 species are shown in bold. Species in italics indicate absence from one Season. Bold values indicate significant ratio of dissimilarity to standard deviation > 0.75.

Species	**Spring** Av. Abund	**Summer** Av. Abund	Av. Diss	Diss /SD	Contrib %	Cum. %
Westralian puller	0.97	1.51	3.87	**1.12**	7.99	7.99
Rough bullseye	0.46	1.37	3.62	**0.82**	7.47	15.47
King wrasse	2.35	1.35	2.79	**1.42**	5.77	21.24
Footballer sweep	2.69	2.21	2.34	**1.33**	4.84	26.08
Black head puller	0.88	0.18	2.28	**1.04**	4.72	30.80
Sand/Silver trevally	1.17	1.67	2.00	**1.21**	4.12	34.92
Banded sweep	0.65	*0.00*	1.62	**1.93**	3.36	38.28
Scribbled chieseltooth wrasse	0.50	0.35	1.54	**0.97**	3.18	41.46
Black spot wrasse	0.60	0.30	1.35	**1.19**	2.79	44.26
Green moon wrasse	0.46	0.35	1.28	**1.22**	2.65	46.91
Yellowtail scad	*0.00*	0.63	1.25	0.44	2.59	49.50
Redbanded wrasse	1.30	1.36	1.25	**1.67**	2.58	52.07
Smooth stingray	0.46	*0.00*	1.18	**1.33**	2.45	54.52
Herring cale	0.46	*0.00*	1.18	**1.33**	2.45	56.97
Maori wrasse	1.11	0.99	1.18	**0.94**	2.43	59.40
Brownspot wrasse	1.20	0.88	1.14	**1.23**	2.35	61.76
Unid wrasse	0.46	*0.00*	1.13	**0.92**	2.34	64.09
Breaksea cod	0.71	0.53	1.13	**1.04**	2.34	66.43
Woodwards reef eel	0.41	0.35	1.08	**1.05**	2.23	68.65
Western talma	0.41	0.12	1.03	**0.97**	2.13	70.79
Senator wrasse	0.58	0.35	0.97	**0.97**	2.00	72.79
Silver drummer	0.46	0.35	0.92	**0.97**	1.90	74.69
Blue lined leatherjacket	0.46	0.35	0.90	**0.95**	1.85	76.54
McCullochs scalyfin	0.76	0.80	0.87	**0.93**	1.81	78.34
Samson fish	0.12	0.35	0.84	**0.94**	1.74	80.08

4.3.2.2. Description of Fish Assemblages Within the Marmion Marine Park

Fish assemblages within Zones of the Marmion Marine Park exhibited considerable variation, with significant differences occurring between Zones that were open or closed to fishing activities (Figure 4.13, Table 4.11). Variations in the abundance of common species contributed to the fish assemblage differences between Zones (Table 4.12). Most of these species were more abundant at sites open to fishing activities, in particular buffalo bream were not present at all in the closed Zone. Additionally there were significant differences between Seasons in the MMP (t=1.4095, p=**0.02**), primarily due to differences in the fish assemblages within the closed Zone (t=1.563, p=**0.02**) but not the open Zone (t=1.1545, p=0.23). Three species (footballer sweep, Samson fish, and the striped stingaree) were not present for one of the Seasons in the closed Zone of MMP (Table 4.13) and along with variations in the abundance of other common species such as silver drummer and silver/sand trevally contributed significantly to the Seasonal differences observed.

The only Category 1 species recorded in MMP were Samson fish and pink snapper, both in low numbers but slightly more abundant during the spring survey in Sites open to fishing activity (Table 4.3.).

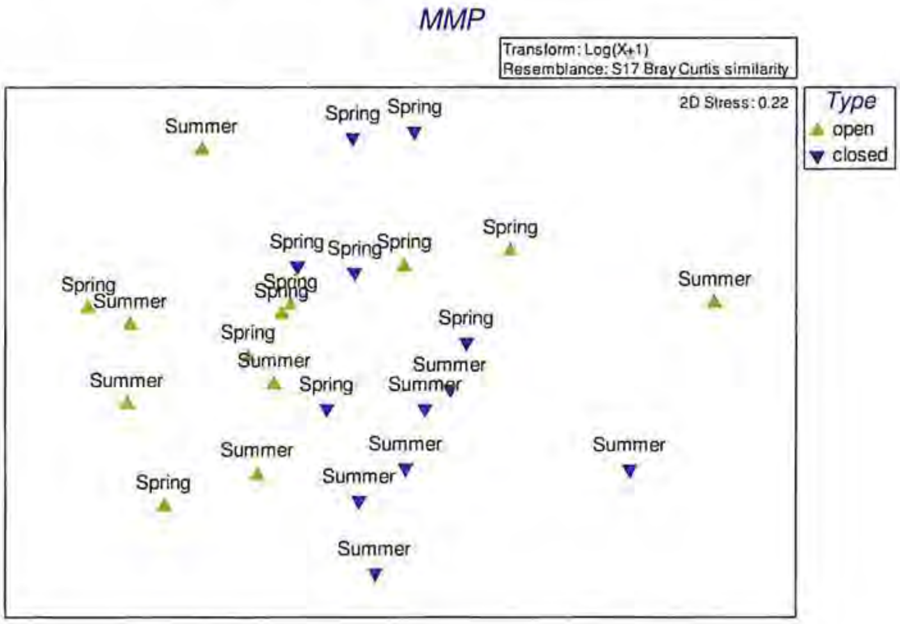

Figure 4.13: Non-metric MDS ordination of fish assemblages observed during SBRUV surveys of Sites within the Marmion Marine Park.

Table 4.11: Results of PERMANOVA pairwise analysis of Zone for both surveys and by Season for MMP. Bold values indicate significance $p<0.05$.

	Both surveys		Spring		Summer	
Comparison	**t**	**p**(perm)	**t**	**P**(perm)	**t**	**p**(perm)
Open, Closed	1.4256	**0.0276**	1.3559	0.087	1.3562	0.0778

Table 4.12: Results of similarities and percentages analysis between Zones within the Marmion Marine Park. Species in italics indicate absence from one Zone. Bold values indicate significant ratio of dissimilarity to standard deviation > 0.75.

Species	Open Av. Abund	Closed Av. Abund	Av. Diss	Diss /SD	Contrib %	Cum. %
Sand/Silver trevally	1.54	0.81	5.51	**1.25**	10.49	10.49
Silver drummer	0.32	0.86	3.13	**0.88**	5.96	16.45
Buffalo bream	0.88	*0.00*	2.96	0.65	5.64	22.09
King wrasse	1.92	1.78	2.62	**1.46**	4.99	27.08
Redbanded wrasse	0.76	0.38	2.61	**1.24**	4.97	32.05
Rough bullseye	0.27	0.50	2.29	**0.89**	4.36	36.42
McCullochs scalyfin	1.26	1.06	2.00	**1.19**	3.80	40.22
Western talma	0.57	0.45	1.82	**1.06**	3.46	43.68
Footballer sweep	0.21	0.41	1.80	0.51	3.42	47.10
Horseshoe leatherjacket	0.24	0.50	1.78	**1.01**	3.40	50.50
Black spot wrasse	0.44	0.55	1.72	**1.10**	3.28	53.77
Southern maori wrasse	0.54	0.77	1.68	**1.20**	3.20	56.98
Smooth stingray	0.43	0.12	1.46	**1.12**	2.78	59.76
Port Jackson shark	0.33	0.15	1.40	**0.83**	2.67	62.43
Woodwards reef eel	0.21	0.40	1.39	**1.02**	2.64	65.07
Herring cale	0.27	0.17	1.36	**0.72**	2.58	67.65
Brownspot wrasse	1.15	1.15	1.16	**1.17**	2.20	69.86
Eagle ray	0.24	0.17	1.15	0.73	2.18	72.04
Banded sweep	0.32	0.17	1.08	**0.86**	2.06	74.10

Table 4.13: Results of SIMPER analysis of fish assemblage differences between Seasons in the closed Zone at MMP. Species in *italics* indicate absence from one Season. Bold values indicate significant ratio of dissimilarity to standard deviation > 0.75.

Species	**Spring** Av. Abund	**Summer** Av. Abund	Av. Diss	Diss /SD	Contrib %	Cum. %
Sand/Silver trevally	1.51	0.12	5.39	**1.77**	10.23	10.23
Silver drummer	1.23	0.48	4.54	**1.06**	8.62	18.85
Footballer sweep	*0.00*	0.81	3.22	0.63	6.11	24.96
Western talma	0.12	0.78	2.82	**1.61**	5.35	30.31
Rough bullseye	0.30	0.69	2.52	**1.08**	4.79	35.10
McCullochs scalyfin	1.27	0.85	2.16	**1.60**	4.11	39.20
Horseshoe leatherjacket	0.65	0.35	2.13	**1.08**	4.05	43.25
Yellow headed hulafish	0.48	0.12	2.01	0.54	3.81	47.06
Redbanded wrasse	0.30	0.46	1.93	**1.00**	3.66	50.72
Black spot wrasse	0.35	0.76	1.67	**1.05**	3.16	53.88
King wrasse	1.84	1.72	1.49	**1.35**	2.83	56.71
Woodwards reef eel	0.35	0.46	1.36	**0.95**	2.58	59.30
Samson fish	0.35	*0.00*	1.29	**0.97**	2.46	61.76
Maori wrasse	0.90	0.65	1.28	**0.93**	2.43	64.18
Striped stingaree	*0.00*	0.35	1.24	**0.97**	2.36	66.54
Senator wrasse	0.46	0.58	1.15	**0.77**	2.18	68.72
Eagle ray	0.12	0.23	1.13	**0.77**	2.15	70.87
Brownspot wrasse	1.13	1.16	1.08	**1.10**	2.06	72.93
Herring cale	0.23	0.12	1.04	0.60	1.97	74.90
Banded sweep	0.23	0.12	1.02	**0.77**	1.94	76.84

4.3.2.3. Description of Fish Assemblages Within Shoalwater Islands Marine Park

Despite an abundance of seemingly good habitat, the Shoalwater Islands Marine Park was noteworthy for the low number of species, diversity and abundance of fish species (Table 4.3.). Species that occurred in reasonably high abundances at RI and MMP such as footballer sweep, rough bullseye, silver drummer, red-banded wrasse and McCullochs scalyfin were either absent or in very low abundance within the SWIMP (Table 4.3.). Three individual Samson fish were the only Category 1 species observed during surveys of this Area. The nMDS plot of the SWIMP fish assemblages by Zone (Figure 4.14.) shows no separation and pairwise tests confirmed there were no significant differences between Zones or Seasons (Table 4.14.). Fish assemblages observed at additional sites in the spring survey with depths >10 m were not different to those in the shallower water survey sites (Figure 4.14.).

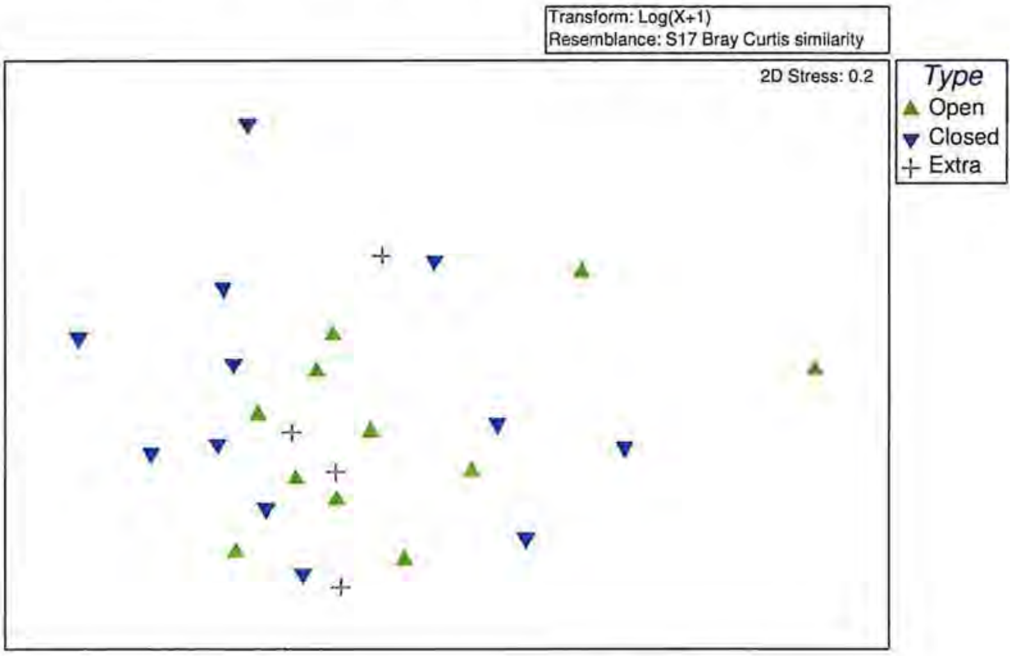

Figure 4.14: Non-metric MDS ordination of fish assemblages observed during SBRUV surveys of Sites within the Shoalwater Islands Marine Park.

Table 4.14: Results of PERMANOVA pairwise analysis of Zone for both surveys and by Season for SWIMP. Bold values indicate significance p<0.05.

	Both surveys		**Spring**		**Summer**	
Comparison	**t**	**p** (perm)	**t**	**p** (perm)	**t**	**p** (perm)
Open, Proposed Closed	1.0721	0.3144	0.64306	0.8684	1.1245	0.2888

4.3.3. Abundance and Size of Focal Species

4.3.3.1 Category 1 Species

Few Category 1 species were observed within the Shoalwater Island and Marmion Marine Parks (Table 4.3). Within the SWIMP a total of three Samson fish were recorded over both Seasons. During the summer survey of the MMP only a single undersized pink snapper (LCF=279 mm) was observed, whilst in the spring survey two small pink snapper (LCF=261 and 242 mm) and seven Samson fish were recorded.

In contrast, 176 individuals belonging to nine Category 1 species were recorded during the surveys of Rottnest Island. The number of species and relative abundance (Total MaxN) of these species varied, although not significantly between Zones and Seasons at RI (Figure 4.15). Of note were the higher total maxN at KR and AB, particularly in spring. Further information on the distribution, relative abundance and size for each of these species around RI is provided below.

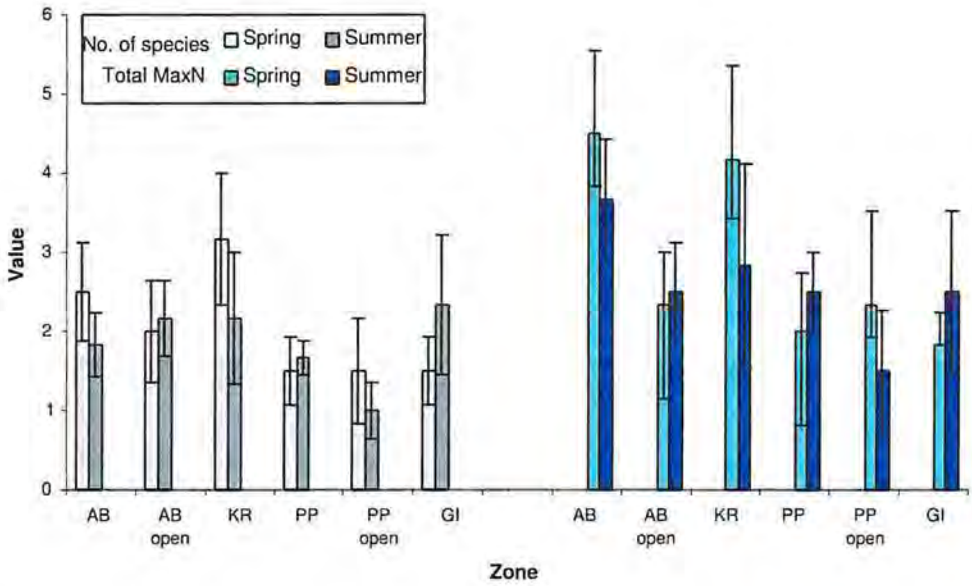

Figure 4.15: Mean number of species and total MaxN for Category1 species in each Zone at RI for each Season, with standard errors.

4.3.3.1.1. WA Dhufish

Dhufish exhibited an inquisitive response to the SBRUV and occasionally fed at the bait. Six dhufish were observed at RI in spring whereas one was observed in summer. Only one of these fish, at 505 mm TL, was above the minimum legal length (MLL) of capture for this species (500 mm). On two occasions during the spring survey an individual of similar size and sex was recorded on consecutive days at the same Site, indicating that they were probably the same fish. Therefore it is likely that only four individuals were actually observed during spring. These included a 397 mm female in the KR Sanctuary Zone, a 466 mm male within the GI Sanctuary Zone, and a 458 mm female and 505 mm male within the PP Open Zone. The sole dhufish seen in the summer survey, a 483 mm female, was observed within the GI Sanctuary Zone. Note that sex is based on dorsal fin filament length (Mackie et al *In Prep*).

The DoFWA Recreational Angler Program (RAP) daily fishing logbook recorded ten dhufish caught in the vicinity of Rottnest Island during the past three years. Six of these were above the minimum legal length and two were above 700 mm in length. These two fish were caught to the west of RI (Smith and Hammond, DoFWA unpubl. Data). Refer to Attachment 1 for further details of the RAP that focused on the shore and near-shore capture of fish around RI. Note that the RAP and SBRUV data are not directly comparable spatially or temporally. In this instance they are regarded as providing complementary, overlapping data for certain fish species.

4.3.3.1.2. Pink snapper

Pink snapper (snapper) were responsive to the SBRUV and usually fed at the bait. Their aggressive feeding sometimes increased the bait plume and attracted other fish to the location. Snapper were more abundant during the spring survey (N=18) when twelve were present at Zones to the north (including five within KR), and six to the south of RI. In contrast, only seven snapper were observed during the summer survey and all except one were present within the KR Sanctuary Zone. Snapper observed during spring were also considerably larger than those present during summer (mean FL 442 mm, range 349 – 609 mm versus mean of 360 mm and range of 298 – 395 mm, respectively). When this length data was converted to TL, eight of the snapper observed during spring were above the new MLL (October 2008) of 450 mm TL for this species (two in KR and PP Open and four in PP). Two of the snapper observed during the summer survey were above the MLL (both in KR).

The DoFWA RAP recorded nine snapper caught near RI during the past three years. These ranged in size up to 480 mm TL. Five of these fish were above the MLL for this species (Smith and Hammond, DoFWA unpubl. data). Five of the snapper observed during the SBRUV surveys were larger than 480 mm TL (FL converted to TL by equation from Wakefield 2006), the largest was estimated at 705mm TL.

4.3.3.1.3. Baldchin groper

Baldchin groper were generally non-responsive to the SBRUV during the current study (whilst noting feeding behaviour has been observed by individuals of this species during other surveys conducted elsewhere, Watson et al 2008). They were only seen at Zones to the north of RI. During the spring survey four individuals, ranging in length from 315 – 478 mm (FL) were recorded. Three of these were above the MLL of 400mm TL for this species (two within AB and one within KR). During summer two individuals were recorded within AB Open, although neither could be measured.

The DoFWA RAP recorded twelve baldchin groper up to 650 mm during the past three years. Eight of these were larger than the largest individual observed during the SBRUV surveys. All except one of these larger fish were caught in waters on the west of RI (Smith and Hammond, DoFWA unpubl. data).

4.3.3.1.4. Blue groper

Blue groper were non-responsive to the SBRUV. Those that were seen were in the nearby vicinity of the SBRUV when dropped, as indicated by the time of arrival being in the first 5 minutes (Table 4.1). Four were observed within Zones to the north of RI during the spring survey. These included a large male within the KR Sanctuary Zone that could not be measured (sex based on colouration), two females within the AB Open and one female within the AB Sanctuary Zone. Two females that could be measured were 370 and 475 mm FL. Neither were above the MLL for this species of 500 mm TL. Only one groper was seen during the summer survey - a female that was 324 mm FL within the GI Sanctuary Zone to the south of RI.

The DoFWA RAP recorded one blue groper during the past three years. This individual was 520 mm in length and caught to the south of Parker Point during November (Smith and Hammond, DoFWA unpubl. data).

4.3.3.1.5. Breaksea cod

Breaksea cod were responsive to the SBRUV and often positioned themselves at the baitbag or approached their reflection in the camera housing but rarely fed. The maximum number of individuals (MaxN) seen at a Site was four on four occasions. They were the most abundant Category 1 species observed during the surveys, comprising 45% of the total number of these species, and were seen on 65% of videos at RI. Breaksea cod were almost equally abundant during spring and summer surveys although numbers varied considerably during replicate days of the surveys (Table 4.15). This was particularly the case within the AB Sanctuary Zone where the largest number of cod was observed in both Seasons. In both Seasons the average length of breaksea cod was greatest within the KR Sanctuary Zone, although ANOVA indicated that maean length differences between Zones were not significant at P = 0.05.

The DoFWA RAP recorded twenty breaksea cod during the past three years. Most of these were captured to the west of RI (Smith and Hammond, DoFWA unpubl. data). A comparison of length distributions shows that breaksea cod sampled by SBRUVs covered a broader size range but were generally smaller than those captured by fishers (Figure 4.16). Larger-sized cod reported in the RAP were often captured in waters west of RI.

Table 4.15: Average length (Avge Lt), with standard errors, of breaksea cod in each Zone at Rottnest Island by Season, with MaxN observed on each day of SBRUV surveys.

	Spring					Summer				
RI Zones	**Avge Lt** (mm)	**Stand Err**	**Day 1** (#)	**Day 2** (#)	**Total** #	**Avge Lt** (mm)	**Stand Err**	**Day 1** (#)	**Day 2** (#)	**Total** #
KR	329.9	31.7	4	3	7	287.6	56.6	1	3	4
AB	260.7	21.7	11	2	13	253.7	17.3	4	10	14
AB Open	261.2	22.6	5	2	7	258.6	20.2	1	5	6
PP	221.3	37.4	3	2	5	228.9	73.6	5	6	11
GI	219.4	7.9	4	3	7	259.2	10.9	4	1	5
PP Open	303.7	58.8	3	2	5	-	-	5	0	5
Total			**30**	**14**	**44**			**22**	**12**	**45**

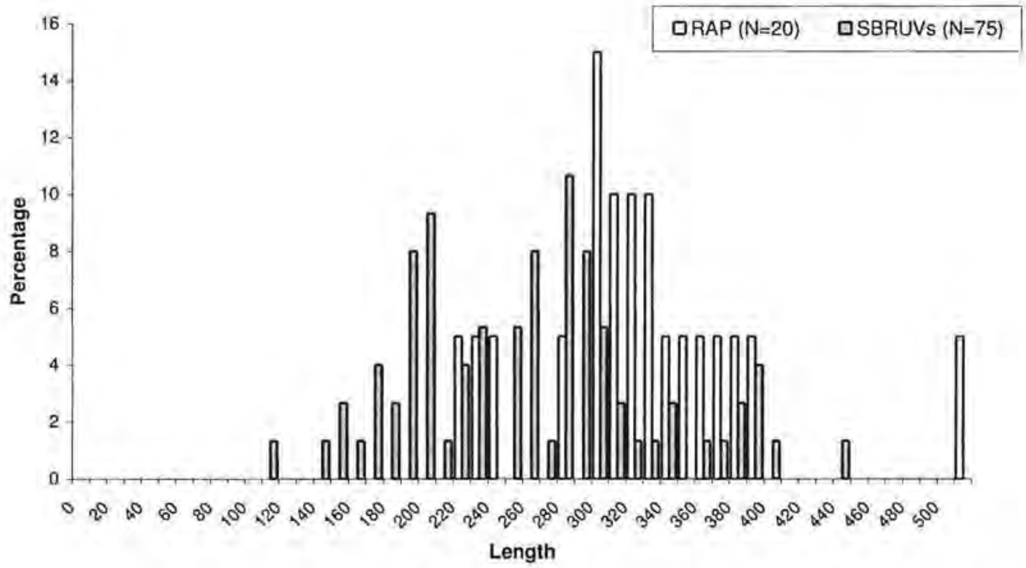

Figure 4.16: Length frequency distribution of breaksea cod from SBRUV surveys and the Recreational Angler Program log book (Smith and Hammond; unpublished data).

4.3.3.1.6. Samson fish

Samson fish were responsive to the SBRUVs often circling a number of times and occasionally attempting to feed. They were the second most abundant of the Category 1 species, with between one and three individuals observed on 42% of videos at RI. Twelve individuals were seen in the NRI Zones during spring (474 – 1254 mm FL) whereas eleven (562 – 1275 mm) were recorded during summer. In contrast, three (985 – 1241 mm) were seen in SRI Zones in spring compared to nine (994 – 1346 mm) in summer. Only one of these fish was below the MLL for this species.

4.3.3.2. Comparative Species

The comparative species were chosen based on the criteria of being;

1. among the most abundant species encountered in the surveys,
2. responsive to the bait and SBRUVs,
3. present in each of the study Areas,
4. occasionally retained by recreational fishers, so potentially influenced by sanctuary zones.

4.3.3.2.1. King wrasse

Individuals of this species were responsive to the SBRUVs and usually fed at the bait. They were the most abundant species recorded in the surveys (Table 4.3) and are occasionally kept by recreational fishers, usually for bait. The RI Zones had higher relative abundances of king wrasse (Figure 4.17) but each Zone was highly variable within (large errors) and between Seasons, though not significantly (Table 4.15). Abundances were not significantly different in each Area but at SWIMP were significantly lower than the northern RI Zones (AB, AB Open, and KR), and PP Open (Table 4.15), while MMP closed was significantly lower than AB and KR.

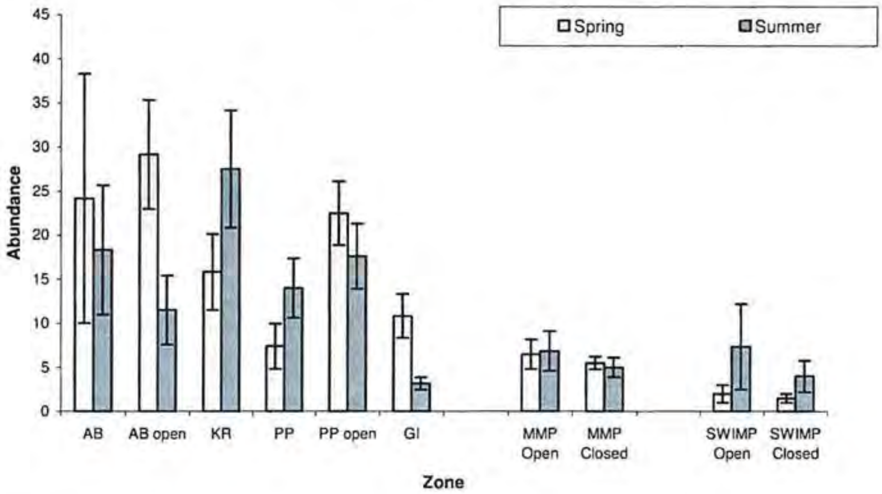

Figure 4.17: Mean relative abundance of King wrasse in each Zone by Season, with standard errors.

Table 4.16: Results of ANOVA for king wrasse abundance by Season and Zone, with Tukey post-hoc pairwise comparison for Zone. Bold values indicate significance of $p<0.05$.

Source	**Sum of Squares**	**df**	**Mean Square**	***F*-ratio**	**p**
Season	65.209	1	65.209	0.494	0.484
Zone	7217.368	9	801.930	6.075	**0.000**
Season X Zone	1969.546	9	218.838	1.658	0.109
Error	13333.024	101	132.010		

Tukey HSD — Matrix of pairwise comparison probabilities:

	MMP open	MMP close	AB	AB open	GI	KR	PP	PP Open	SWIMP Close	SWIMP Open
MMP open	1.000									
MMP close	0.084	1.000								
AB	0.084	**0.037**	1.000							
AB open	0.134	0.064	1.000	1.000						
GI	1.000	1.000	0.100	0.158	1.000					
KR	0.067	**0.029**	1.000	1.000	0.080	1.000				
PP	0.999	0.991	0.382	0.508	1.000	0.329	1.000			
PP open	0.290	0.158	1.000	1.000	0.329	1.000	0.752	1.000		
SWIMP close	0.988	0.999	**0.003**	**0.006**	0.981	**0.002**	0.752	**0.020**	1.000	
SWIMP open	0.994	1.000	**0.004**	**0.007**	0.990	**0.003**	0.802	**0.023**	1.000	1.000

Average lengths at each Zone were usually similar between Seasons, except within AB open and GI where larger fish were observed during summer (Figure 4.18), which corresponds with the lower abundances at these Zones (see above). There were also significant differences in the mean length of king wrasse between Zones in each Season (Table 4.16), driven mainly by the larger size and lower abundance of King wrasse within the MMP and SWIMP Zones compared to those within the RI Zones (Figures 4.18 and 4.17, respectively). The schools of female and juvenile king wrasse often seen at RI were absent from AB Open and GI in summer and the other Areas surveyed in this study (MMP and SWIMP).

The length distribution of King wrasse recorded in the DoFWA RAP consists of only the larger individuals that were detected in low numbers by the SBRUV surveys (Figure 4.19; Smith and Hammond, DoFWA unpublished data). This is most likely due to recreational fishing methods only capturing individuals above 250mm.

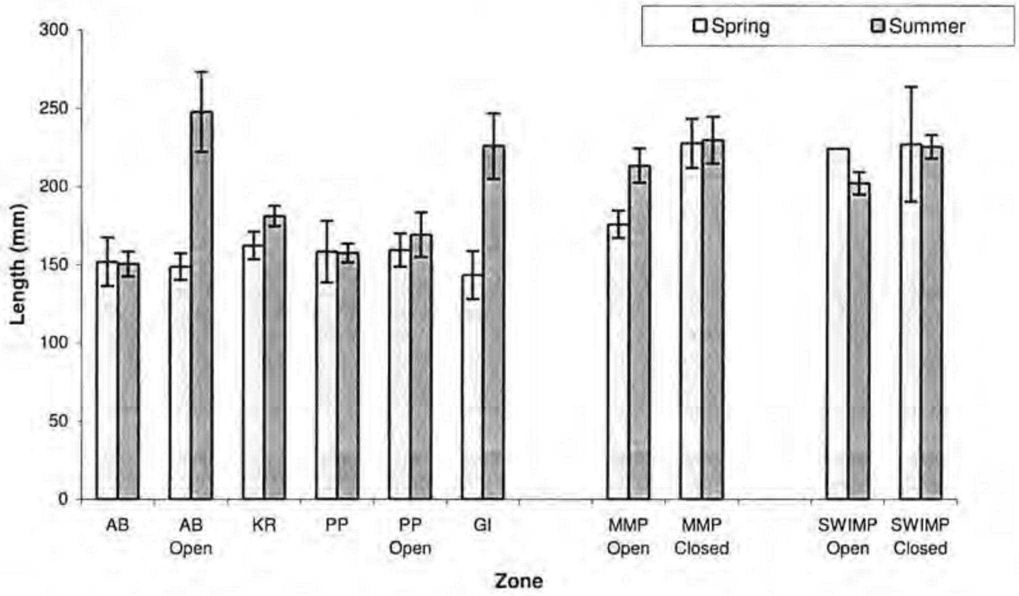

Figure 4.18: Mean length of king wrasse (mm) in each Zone by Season, with standard errors.

Table 4.17: Results of ANOVA for king wrasse length (mm) by Season and Zone, bold values indicate significance P<0.05.

Source	Sum of Squares	df	Mean Square	*F*-ratio	p
a) Overall					
Season	26895.634	1	26895.634	6.939	**0.009**
Zone	205888.722	9	22876.525	5.902	**0.000**
Season X Zone	185612.965	9	20623.663	5.321	**0.000**
Error	2100823.159	542	3876.057		
b) Spring					
Zone	178607.427	9	19845.270	4.583	**0.000**
Error	1030661.177	238	4330.509		
c) Summer					
Zone	275357.504	9	30595.278	8.691	**0.000**
Error	1070161.982	304	3520.270		

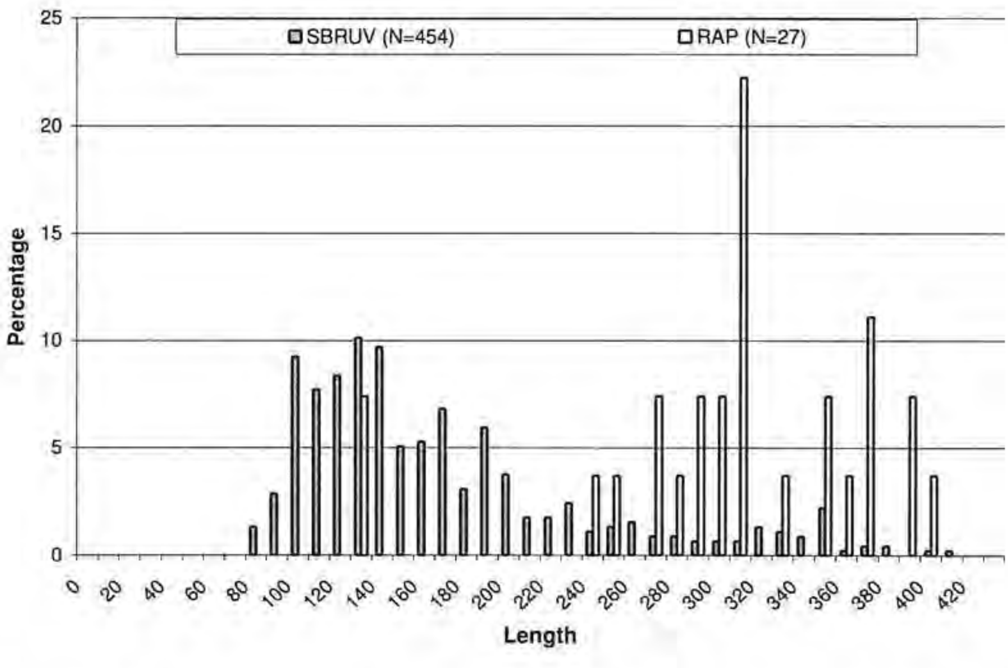

Figure 4.19: Comparison of length frequency data for King wrasse at Rottnest Island from the SBRUV and RAP surveys (Smith and Hammond unpublished data).

4.3.3.2.2. Brownspot wrasse

Brownspot wrasse were responsive to the SBRUV and usually fed at the bait. They can reach a size of 490 mm and are occasionally kept by recreational fishers so may be influenced by the establishment of MPAs. This species had low relative abundances, compared to King wrasse, with the numbers observed in each Zone ranging from six to nineteen during both Seasons (Table 4.3.). The abundances were similar between Seasons, and Zones (Figure 4.20.) with a significant difference between zones (Table 4.18.). Pairwise tests showed relative abundance of brownspot wrasse in PP Open was significantly lower than AB Open and KR.

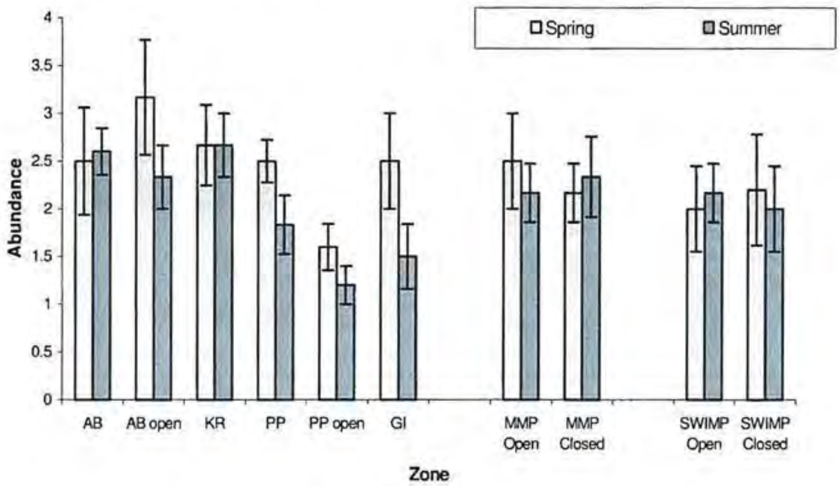

Figure 4.20: Mean relative abundance of brownspot wrasse in each Zone by Season, with standard errors.

Table 4.18: Results of ANOVA for brownspot wrasse abundance by Season and Zone.

Source	Sum of Squares	df	Mean Square	F-ratio	p
Season	2.226	1	2.226	2.093	0.151
Zone	21.547	9	2.394	2.251	**0.024**
Season X Zone	6.107	9	0.679	0.638	0.762
Error	107.429	101	1.064		

There was also little consistent difference, none significant, in the mean lengths of brownspot wrasse within each Zone and Season (Figure 4.21). The maximum recorded length for this species by SBRUVs was 422 mm TL at KR.

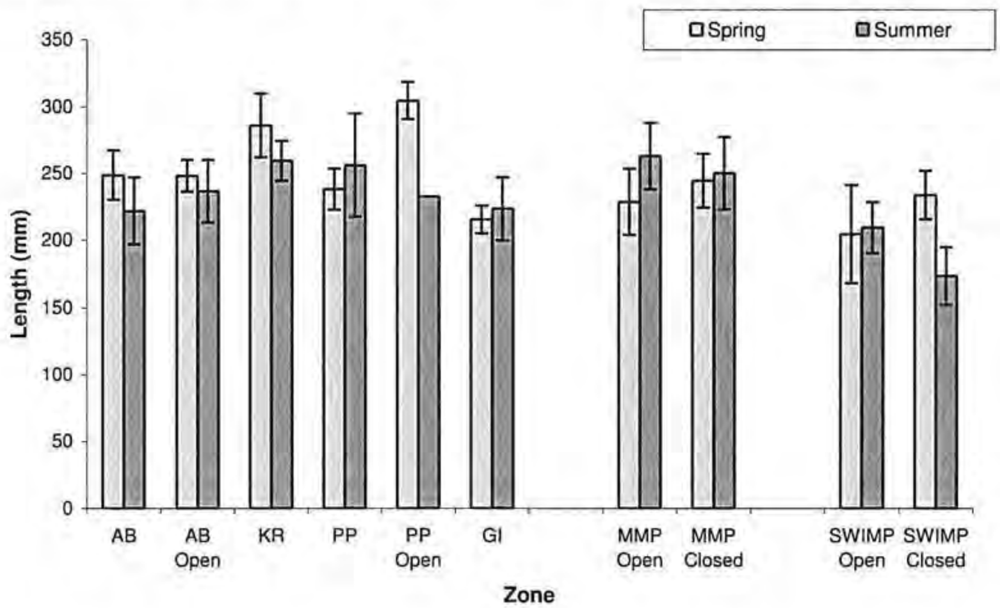

Figure 4.21: Mean length of brownspot wrasse (mm) in each Zone by Season, with standard errors.

4.3.3.2.3. Silver/Sand trevally

Data for these species were grouped due to their similarity and problems with identification on video. They were responsive to the SBRUV and usually fed at the bait. Trevally are a regular target of recreational anglers and were encountered in all Areas. They are a schooling species that have potential for an underestimate of abundance.

The mean abundances in each Zone (Figure 4.22) were not significantly different in spring but were in summer (Table 4.18). Pairwise investigation showed AB had significantly higher abundance of the species than PP, PP Open and MMP Open was significantly higher than MMP Closed. These results at MMP are influenced by a school of at least 40 individuals that was seen in the open Zone on 2 occasions.

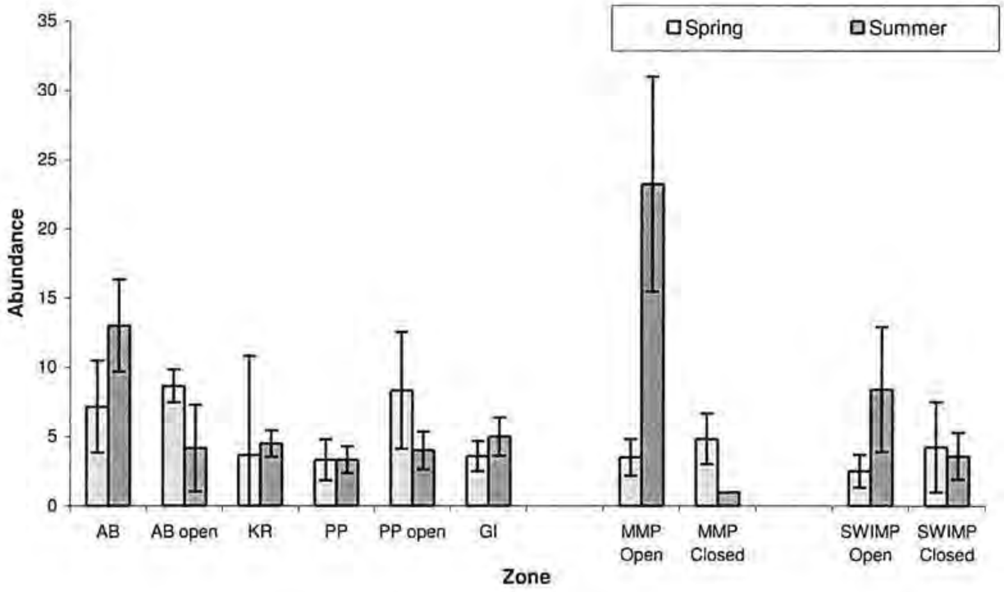

Figure 4.22: Mean abundance of silver/sand trevally by Zone in each Season, with standard errors.

Table 4.19: Results of ANOVA for sand/silver trevally abundance by Season and Zone.

Source	Sum of Squares	df	Mean Square	*F*-ratio	p
a) Overall					
Season	109.901	1	109.901	2.135	0.147
Zone	959.082	9	106.565	2.070	**0.039**
Season X Zone	1190.665	9	132.296	2.570	**0.010**
Error	5199.548	101	51.481		
b) Spring					
Zone	409.062	9	45.451	1.039	0.423
Error	2230.381	51	43.733		
c) Summer					
Zone	1745.017	9	193.891	3.265	**0.003**
Error	2969.167	50	59.383		

At SWIMP and MMP the mean lengths of silver/sand trevally were not significantly different (Figure 4.23, Table 4.20). However, there was distinct spatial and seasonal variation in the lengths of trevally in Zones around RI. In summer AB had significantly smaller silver/sand trevally than all other RI Zones and in spring PP Open was significantly smaller than all others while KR and AB were significantly larger than all except each other. As with king wrasse the higher abundance at AB in summer and PP Open in spring corresponds with a smaller average length at these zones than others. This illustrates the variability and mobility of these species as the AB zone changed from having low abundant larger fish, in spring, to higher abundance but smaller fish, in summer. Also of note were the seasonal differences in the lengths of fish within the smallest cohort at RI (Figure 4.24), which may be due to the combined lengths of the 2 species.

The length distribution of skippy/sand trevally recorded in the DoFWA RAP is similar to that of the SBRUV surveys (Figure 4.25; Attachment 1, Smith and Hammond, DoFWA unpublished data), with more of the larger-sized fish (>400mm) recorded by the RAP and smaller fish (<200mm) by the SBRUVs.

Table 4.20: Results of ANOVA for length of silver/sand trevally by Season and Zone, bold values indicate significance P<0.05.

Source	Sum of Squares	df	Mean Square	*F*-ratio	p
a) Overall					
Season	2079.245	1	2079.245	0.825	0.365
Zone	231291.800	9	25699.089	10.191	**0.000**
Season X Zone	304278.625	9	33808.736	13.407	**0.000**
Error	799401.025	317	2521.770		
b) Spring					
Zone	575407.499	9	63934.167	27.148	**0.000**
Error	317922.592	135	2354.982		
c) Summer					
Zone	424389.815	9	47154.424	17.824	**0.000**
Error	481478.433	182	2645.486		

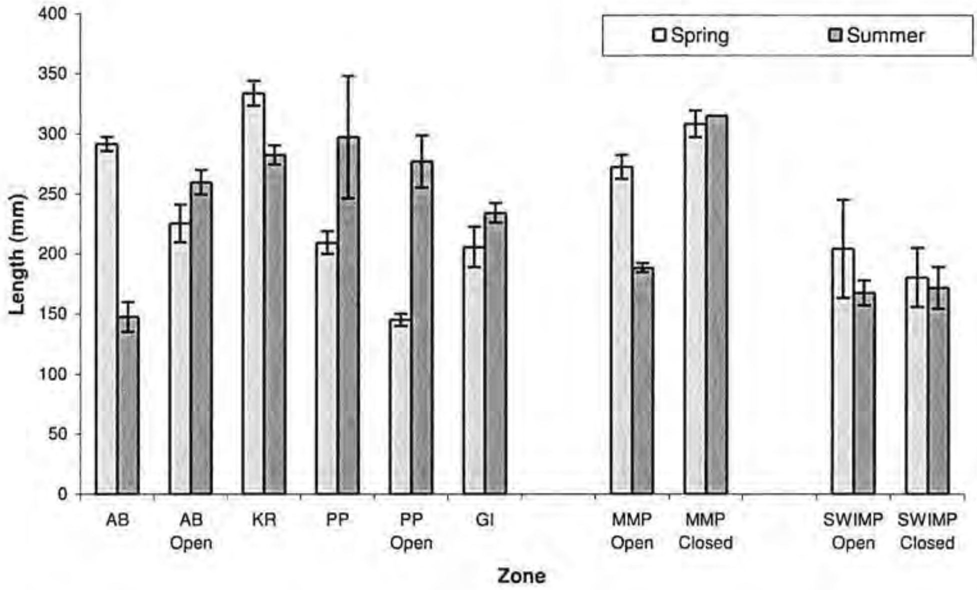

Figure 4.23: Mean length (Caudal fork, mm) of silver/sand trevally in each Zone by Season, with standard errors.

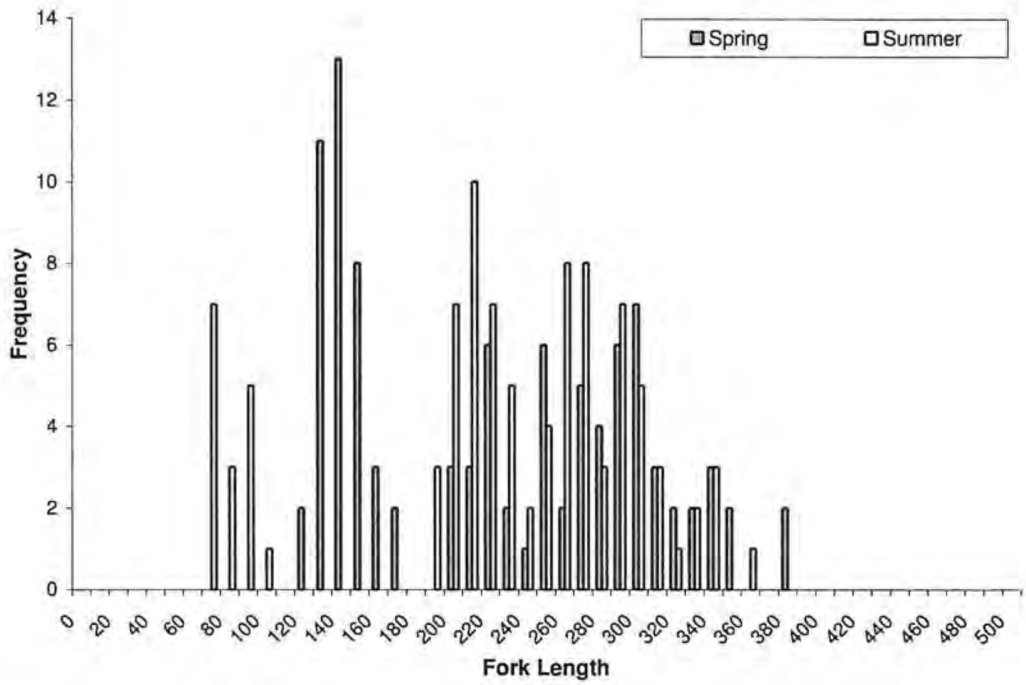

Figure 4.24: Length frequency (Caudal fork, mm) of silver/sand trevally for each Season at Rottnest Island from SBRUV surveys.

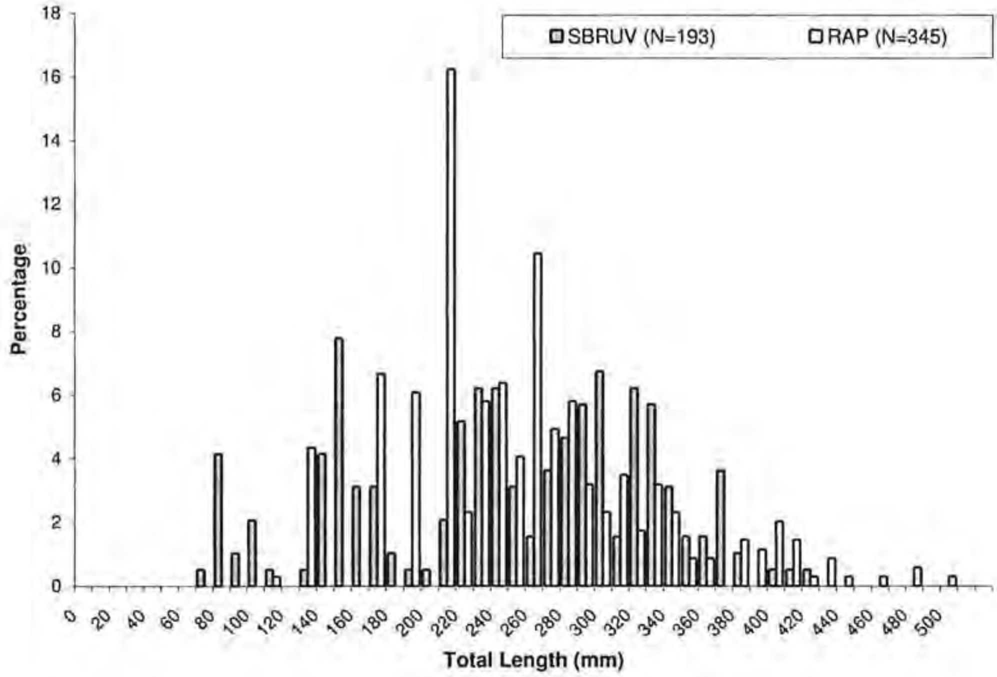

Figure 4.25: Comparison of length frequency data (TL, mm) for silver/sand trevally from the SBRUV and RAP surveys (Attachment 1) at Rottnest Island.

4.4. Discussion

This study describes and compares fish assemblages in and around coastal marine parks within Perth metropolitan waters. It focuses on species that are considered to be at risk from fishing pressure. As this study is also a forerunner to a long-term monitoring program, the sampling methods and design have been reviewed in order to maximise the efficiency and effectiveness of future surveys.

4.4.1. Sampling Design

Baited remote underwater video systems provide a standardised sampling technique, unique visual imagery of habitat and fauna plus the ability to go to depths with greater replication without the need to use divers. Stereo BRUVS enable additional length measuring capacity. However, like all sampling methods the data obtained from BRUVS can be biased. Potential sources of bias were examined during the current study in order to improve the quality of the data arising from this and future surveys. It was determined that:

- The response of fish species to the SBRUV and bait affected the likelihood that they are recorded during a particular survey. As a consequence, each Site needs to be surveyed on replicate days in order to properly describe the species assemblage at that location. During this study surveys were completed on two replicate days. Further investigation is required to determine if this is sufficient for full description of the local fish assemblage including crepuscular and nighttime sampling. Increased temporal and spatial replication will also reduce the impact of 'dud' deployments.

- The direction and angle in which SBRUV units are deployed do not generally impact conclusions drawn from the data (noting that assessment of this during the current study was not exhaustive). This is useful because there is usually little control over the way a unit lands on the seabed. Nevertheless, analysis of video indicate that a camera field of view over sand can provide different data from one over reef or angled up into the water column, even if the location is exactly the same. Video footage from future surveys should therefore be categorised as described in this study to confirm reliability of the data in future surveys.

- The survey sites chosen for this study were far enough apart not to be adversely affected by overlap of bait plume due to current strength. Current strength and the associated likelihood of a fish encountering the bait plume also did not appear to have a significant impact on fish abundances and other data obtained from the surveys. Swell height also had minor influence on data obtained during the surveys, although observations of video footage showed that large swell has a large impact on visibility and fish behaviour in these shallow water Sites. Nevertheless, the impact of swell on data analyses is minor, simply because safety and equipment concerns prohibit surveys on days when the swell is too large.

- Sixty minutes may not be sufficient time to properly survey Category 1, uncommon and unresponsive species. However, longer survey times require a change in camera recording methods to hard disk drive or memory stick cameras which allow 2-4hours to be recorded but these have associated additional costs of processing, footage capture and storage. Increased replication is also likely to enhance the likelihood of encountering these species. Kleczkowski *et al.* (2008) used 30-minute time periods during video surveys of fish fauna at Rottnest Island, recording 59 species compared to the 82 species observed during the current study.

- The time of year when surveys are conducted has a large impact on survey data, particularly for mobile and Category 1 species. During the current study the number and relative abundance of Category 1 species were higher in September ('spring') than they were the following January ('summer'), indicating that spring is preferable if given the choice. It is certainly the case that the timing of future surveys will need to be kept consistent.

Other factors to consider for future surveys include:

- Appropriateness of current survey design. Focal species chosen for more detailed assessment of distribution, abundance and length were either smaller-sized and abundant at each site, hence considered a useful indicator of environmental conditions, or larger-sized, less common species that were a direct indication of the effects of fishing pressure. Two species, breaksea cod and trevally (the latter a mix of two similar looking species), were both abundant at most RI sites and potential indicators of fishing activity. However a focus on the larger, key recreational species was not in keeping with the aim to evaluate the effectiveness of sanctuary zones in the protection of these species. This is because the sanctuary zones are likely to be too small for the seasonal movements of the larger species. These species were also too rare within the MMP and SWIMP to provide useful data for these Areas. In addition, apart from the long-established KR Sanctuary Zone (less than 10m), the sanctuary zones at RI lacked sufficient habitat at appropriate depths (> 10 m) to be surveyed without overlap into Zones open to fishing activities.

As such, and given the DoFWA priority towards these Category 1 species and the costs of undertaking these surveys (as detailed below), it will be better to stop or greatly reduce sampling effort within the MMP and SWIMP in future surveys, instead focussing on improving surveys around RI. This would require a reassessment of current sampling strategy, with inclusion of the relatively large and deep West End Demersal Zone (noting the increased difficulty of sampling at this location), since data from the Recreational Angler Program indicate that larger fish are more abundant there. Deeper reefs in the vicinity of RI that are outside sanctuary zones but known to hold individuals of these Category 1 species may also need to be included. At the same time many of the current sites may be surveyed with less replication in order to monitor distribution and abundance of smaller, responsive species that are reliably and quickly attracted to the SBRUV units although to account for inherent variability and allow for comparisons with this and other surveys the replication is recommended to be maintained at current levels.

- Cost of undertaking the surveys, which vary considerably if the aim is principally to describe community composition within each Area or to focus on the less common Category 1 species. If the former, the costs can be reduced considerably by shortening the survey period to thirty minutes, since the data show that this is sufficient to record the common species that are the main components of local fish assemblages. In addition, only one day of survey is needed per Area to record these common species. Therefore, if three sites are surveyed within each Area as per the current study, then the cost of an annual survey around Rottnest Island would be approximately $4000 (Table 4.21).

However, if the aim is to monitor the Category 1 species the comparable cost of Rottnest Island would be approximately three times this value due to the longer deployment time required and need for replication on consecutive days (Table 4.21).

Table 4.21: Components and approximate costs of a) a basic fish assemblage survey at Rottnest Island and b) a survey focussed on detecting Category 1 species.

Component	**Units**	**Rate**	**Cost**
a) Basic survey			
Consumables (Bait, fuel, tapes)			$300
Field days x 3 personnel	1	300	$900
Analysis (@1.5hrs/site)	4	210	$820
Writeup	10	210	$2050
Total			**$4070**
b) Category 1 species survey			
Consumables (Bait, fuel, tapes)			$1700
Field days x 3 personnel	6	300	$5400
Analysis (@1.5hrs/site)	15	210	$3075
Writeup	15	210	$3075
Total			**$13250**

4.4.2. Fish Assemblages in Each Area

Fish assemblages differed considerably between Areas and within each Area they also exhibited some seasonal differences. Not surprisingly, the number of species and relative abundance of these within the coastal MMP and SWIMP were lower than the offshore Areas around RI. Similar patterns of lower coastal diversity found by Hutchins (1994) were determined to be due to the influence of the Leeuwin current on the fish assemblages at RI (Hutchins and Pearce 1994). These shallower coastal Areas also appear to be of relatively minor importance to the larger Category 1 species than are the RI locations. Additionally, king wrasse, the most abundant species at 24% of the total MaxN and one of the chosen comparative species, were in higher abundance and lower mean size at RI than these coastal marine parks.

The reduced number of species and relative abundance of fishes throughout the SWIMP was surprising, however, given the well-flushed oceanography and broad extent of rocky reef that appears favourable to the species present in the surveys. The fact that species not targeted by anglers were also at low numbers indicates that fishing pressure is not a major contributing factor to the diminished fauna in this Area. Rather, it is possible that predation by sea lions, little penguins, cormorants and pelicans keep the local fish fauna at reduced levels since the numerous small islands located within the SWIMP are a haven for these species. Klomp and Woller (1988) showed that little penguins were opportunistic and their diet included reef fishes such as common bullseyes and juvenile buffalo bream.

The MMP has been established long enough (since 1999) for the predicted benefits associated with a sanctuary zone closed to fishing to be evident for some species. However, whilst there were clear differences in fish assemblages between open and closed Zones in this Area, it was found that the relative abundance of many species were at higher levels in locations open to fishing. One exception was silver drummer, which were in higher abundance in the closed zone, as also found by Ryan (2007), but this is unlikely to be an effect of fishing pressure, as they are rarely targeted by recreational fishers, and more likely due to habitat differences as the similar species, western buffalo bream, were only found at one particular site in the open Zone. Certainly, the relatively small size of the proposed closed zone (the Lumps Sanctuary Zone) and the lack of suitable reef habitat is unlikely to provide sufficient protection to the larger, more mobile, key recreational species. Apart from the influence of one large school of trevally there were no differences in the abundance or size of comparative species between Zones. Poaching may be another issue, since an illegally set rock lobster pot was observed in the closed Zone whilst surveys were being conducted.

Apart from Kingston Reef Sanctuary Zone the sanctuary zones around Rottnest Island have not been established long enough for any benefits of protection on fish communities to be noticeable, as time periods of at least 3 years have been suggested (Halpern & Warner, 2002). In the case of many Category 1 species this may never be possible because of the relatively small size of suitable habitat encompassed by the sanctuary zones. Nevertheless, the surveys revealed differences in fish assemblages around the island. Those within KR were distinct, perhaps as a consequence of the long-term closure to fishing as the mean size of breaksea cod was slightly higher, as also found by Kleczkowski et al (2008), plus trevally were larger in summer and the only adult male blue groper was observed in this Zone. However, as no overall higher number of species or relative abundance of Category 1 species was observed and there were no other differences in the mean sizes or abundances of the comparative species it could be that a sanctuary zone of this size is only beneficial to species that are assumed to be relatively site attached, such as breaksea cod. Alternately, it could simply be because this Zone is spatially isolated and has a unique environment compared to the other Zones. For example, the KR fish assemblages were most similar to nearby AB and differed mostly with those observed within the GI Sanctuary Zone located towards the western end of Rottnest Island. This was due mainly to the greater abundance of species such as footballer sweep, pullers and rough bullseye within the GI, which is more exposed to oceanic conditions than are the more sheltered waters of the KR where species of wrasse were more prevalent.

There was also a noticeable difference between fish assemblages to the north and south of Rottnest Island, albeit with considerable overlap in the data for some Zones such as AB Open and PP. These differences were consistent between seasons and as with the above comparison mainly due to a greater abundance of common species, such as king wrasse on the north and footballer sweep on the south. North-south differences have also been reported in other studies of fish species at Rottnest Island, particularly the presence of tropical species located in the shallow (<15m), protected areas found along the southern side of the island (Hutchins 1979). The current study did not record the same abundance and number of tropical species as it did not survey these same areas due to inaccessibility by boat in spring but did observe a number of adults for tropical species such as the moon and green moon wrasse (*Thalassoma lunare* and *T. lutescens*) in the deeper depths (10-15m) of the south RI Zones, as reported by Hutchins (1991).

Differences between Zones at RI were consistent between both seasons but greatest during the summer surveys. Seasonal changes in distribution and abundance were particularly distinct among the Category 1 species, which in most cases were more abundant during spring. This was particularly true for dhufish, pink snapper, baldchin groper and blue groper. Although generally considered to be a sedentary species, tag and release data indicate that dhufish may make limited onshore-offshore movements, particularly during spring prior to spawning (Mackie *et al.* in prep). Similarly, the larger-sized and more abundant pink snapper that were observed during the spring surveys may have been en route to Cockburn Sound where they aggregate to spawn from October (Wakefield 2006).

The variability in fish communities with both Season and Zone is illustrated by the abundance and size of the 3 comparative species at RI, which varied considerably. Overall there was a slightly lower abundance of king wrasse in the GI Zone and brownspot wrasse in the PP Open zone, as was indicated by the north -south differences. In general Zones with a higher abundance of king wrasse or trevally also had a lower mean size due to the influence of larger schools with smaller fish. Although not directly comparable due to different sampling areas and times this study also provides complimentary species length information to that obtained by the RAP recreational fishing logbook (Smith and Hammond 2008). Each method has its advantages and sampling biases but give comparable results.

The information obtained during this study provides baseline data and sampling protocols for future surveys. It is recommended that if the aim is to focus on the key recreational species then most sampling effort should occur around Rottnest Island in winter to spring, with additional, deeper water sites included, where Category 1 species are more likely to occur, and additional study conducted into the effects of time of day and survey duration on the fish assemblage recorded be investigated to help refine the methods.

5. General Discussion

The ecosystem-based approach to fisheries management (EBFM) ensures consideration of the diversity and abundance of the biota, their associated habitats and trophic interactions, resulting in a more thorough understanding of the complex interplay between marine species and their environment. As such EBFM contributes to the implementation of ecologically sustainable development (ESD) for Australian fisheries. Key species found within the shallow water reef communities of the Swan region include recreationally and commercially important species such as the Western Rock Lobster, WA dhufish and Blue Groper. In order to distinguish between potential sources of change (e.g. human *vs* environmental) to marine communities and to specific targeted species, sites were located in areas where impacts from human disturbances were limited, i.e. 'no-take' sanctuary zones and adjacent areas where fishing pressures are present. The Department of Fisheries' studies reported herein provide baseline information on the biodiversity and community structure of three Marine Reserves located within the NRM Swan region, i.e. Marmion Marine Park, Shoalwater Islands Marine Park and Rottnest Island Marine Reserves. Ultimately the goal was to provide information that would allow development of effective and efficient resource condition targets (RCTs). Although, in most cases, it is too early to use the results of these studies as a proxy for change in the marine environment (due to protection from fishing), the continued collection of these data will allow the production of metrics that "describe" biodiversity and ecosystem structure in a pragmatic and measurable way thus providing a good measure of environmental change in the future.

The aim of this project was to develop a robust, long-term monitoring program within the NRM Swan. This was achieved by addressing the following objectives:

2. Provide baseline seasonal quantitative descriptions of shallow-water fish and benthic communities at multiple sites (in marine parks)
3. Develop cost effective, robust monitoring methodologies to be used in a long-term monitoring program that is able to detect changes to these communities through time

5.1. Benthic Communities

This study provided baseline quantification of benthic communities, flora and fauna, in three marine reserves within the Swan region. No effects of protection from fishing were detected in either the flora and fauna categories, invertebrates or kelp holdfast investigations. Some indicators (flora and fauna categories, invertebrates) of location differences were drawn from the analyses, however as this project is based on only one year of data these results should be used with caution. Most importantly it appears from this study that any shifts in the benthic communities will take longer to be seen than for either the Western Rock Lobster or fish communities (see subsequent sections), and in this respect long-term monitoring in the vicinity of decades is probably necessary to measure change in these communities as a result of protection.

5.1.1. Future Recommendations

It is recommended that algal functional groups (as surrogates to species) be used as future indicators of change to the benthic communities, as they require minimal taxonomic expertise. These functional groups are identifiable from close-up (< 1m) digital imagery taken of the substratum, precluding the need to remove vast amounts of material from the marine sites. Digital video footage should be the preferred method of image collection as dive times are less, allowing greater sample replication. Additional collection of digital still images (photoquadrats) every five years will ensure that video footage continues to be the most effective method of capturing information regarding percentage cover of flora and fauna categories. Baseline seasonal information should continue to be collected for the next few years, whereafter (depending on results) sampling may be able to be reduced to annually (e.g. in summer only). The quadrat method should be used to quantify the invertebrate community coupled with testing the effect of lower order taxonomic identification. For all sampling regimes, at each location, replication of transects should be increased.

5.2. Lobster

This project produced preliminary baseline data on western rock lobster relevant for their use as a biological indicator throughout the Swan region, both in the northern and southern regions of coastal waters and offshore at Rottnest Island. Further this project highlighted the importance of a multi-disciplinary approach when studying these animals. The combination of a potting program, utilising several pot types, with underwater visual census (UVC), provided a far more comprehensive assessment of the lobster population, which is essential when using a species such as this as a biological indicator or when assessing the effectiveness of closures in marine parks. It provides baseline data on lobster abundance, size composition, as well as growth rates and size at maturity.

5.2.1. Future Recommendations

Annual surveys would be more cost effective than multiple studies through the year, and provide a standardised sampling period which would be consistent between years. Monitoring in October or November has several potential advantages, including: sampling prior to the commencement of the lobster fishery and hence any extraction; remove biases associated with sampling during the season such as a shift in effort between years; increase the power of any potting analysis, while being cost effective; and also allow tagging of lobsters prior to any migrations ("whites" migration in November) to look at movement patterns of all lobsters within the marine reserve. Future monitoring should continue to include the multidisciplinary sampling approach described for this project. This approach will enable a more holistic examination of the size distribution dynamics and more robust understanding as to potential changes in abundance. There should be continued use of different pot types so that potting captures a more representative sample of the size distribution of lobsters within the marine reserves. Replication of potting and UVC transects should be increased as this would greatly increase the power of any statistical analysis and allow more certainty to be given to any comparison of paired open and closed areas. Tagging should be included as valuable information of animal movement and growth information can be collected.

5.3. Fish Communities

This study provides a baseline description and comparison of the fish assemblages in and around the marine parks within Perth metropolitan waters, focusing on Category 1 species (species that are considered at high risk of over fishing and hence good indicators of fishing pressure). The use of stereo BRUVS to study fish communities not only provides unique visual imagery of habitat and fauna without the need for divers but also important quantitative data on relative abundance from a standard sampling technique and length measuring capacity. During this study it was determined that the response of fish species to the bait and SBRUV unit affected the likelihood that they were recorded during a particular survey. This can be mitigated by increased replication of survey days at sites (i.e. > 2 days) and also increasing spatial replication. It was found that increased replication and sixty minutes duration may not be sufficient to detect the numbers of Category 1 species required to monitor the effectiveness of the sanctuary zones, as many are uncommon in the Metropolitan area and some, such as Blue Groper, are unresponsive. Breaksea cod is the only Category 1 species that was recorded in sufficient numbers and may be provided some degree of protection by the established sanctuary zones at RI. Consideration needs to given to these factors should Category 1 species be the indicators of choice. This is in contrast to only 30 minutes of video footage being required to sample sufficient numbers of the more abundant and responsive species, such as the trevallies, king, brownspot and southern maori wrasse, although these species are less targeted by fishers.

5.3.1. Future Recommendations

The complete absence or low abundance of Category 1 species in the coastal MMP and SWIMP Marine Parks along with the small size of sanctuary zones creates doubts as to the effectiveness of the sanctuary zones for these key recreational and commercial species. In addition, although the Rottnest Island sanctuary zones are considerably larger they do not appear to provide sufficient habitat at appropriate depths (> 10 m) for many of the Category 1 species, which only occurred in these areas in low abundances and on a seasonal basis. As monitoring continues these new sanctuary zones may prove to only be suitable for the smaller and presumably site attached Category 1 species, such as Breaksea cod. Even the longer-established Kingston Reef sanctuary zone failed to show higher abundance, numbers of species or size of many Category 1 species than other areas surveyed. Given the priority toward key recreational species it is therefore recommended that unless larger sanctuary zones are established within the MMP and SWIMP the sampling effort be reduced in these areas with a greater focus on improving surveys around RI. This would involve the inclusion of additional sites in the relatively large West End Demersal Zone and in deeper water habitat outside of this where the Recreational Angler Program and other evidence, from divers and fishers, indicate Category 1 species occur regularly. At the same time the coastal sites may be surveyed with less frequency (not annually) in order to monitor abundance, diversity and size of common species that are reliably and quickly attracted to the SBRUV units. Further it is recommended that future surveys are undertaken in Winter/Spring, as this was when greater abundance of Category 1 species was recorded in the shallower areas of RI, and be of at least 1 hour duration to allow comparison and account for inherent variability. Investigation into fish assemblage results from longer sampling durations and various times of day should also be included in future assessment of methods.

5.4. Conclusions

The above studies have provided baseline quantitative data on benthic and fish communities (in particular Category 1) and on Western Rock for three marine reserves within the Swan Region. They have tested different methodologies and made recommendations on these that will enable future quantification of changes to these communities. Most importantly they have also identified problems with the sizes of some of the current sanctuary zones in terms of their effectiveness of providing protection and hence stock enhancement to the important Western Rock Lobster and Category 1 fish communities. Another notable issue associated with Marmion Marine Reserve was illegal fishing. Illegal potting was recorded (and reported) within the closed zone of the marine reserve during this study. These results highlight the need for future studies such as these to not only monitor changes to these communities but also to provide empirical evidence as to their effectiveness and as a result allow better informed marine planning decisions. It is clear from these results that the objectives of a sanctuary zone (i.e. what is it aiming to protect) are fundamental considerations prior to their design. Undertaking scientific surveys and gathering quantitative baseline data in areas being considered for protection prior to gazettal would enable the establishment of sanctuary zones with clear management objectives.

The Swan Catchment Council is in the position to lobby State Government on the need for monitoring programs such as those described herein. Management bodies need to make marine planning decisions based on scientific research for there to be effective gains in the ecological sustainability of these important resources. As such careful consideration needs to be given as to the size, design and location of marine reserves if they are to provide any gains in the protection of the marine ecosystem.

Further, networks of marine reserves along the Western Australian coast would ensure there is consideration to different dispersal and migration distances of species, as a result there would be cumulative effects of each locations strengths whilst at the same time fulfilling the goals of different stakeholders.

However, the success of any marine reserve can only be measured by long-term (decadal) monitoring programs. It is suggested that the benefits of State-wide monitoring with NRMs working together would be more powerful, i.e. helping with marine conservation, than any one NRM working in isolation.

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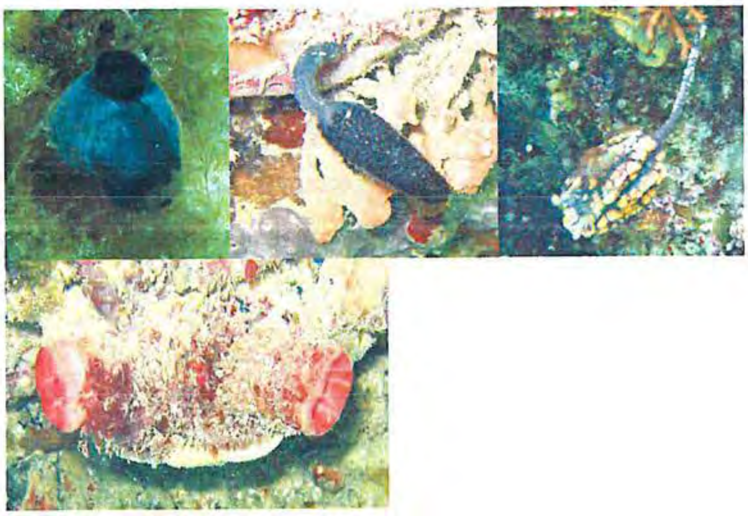
Appendix 1:

Invertebrate field sheet

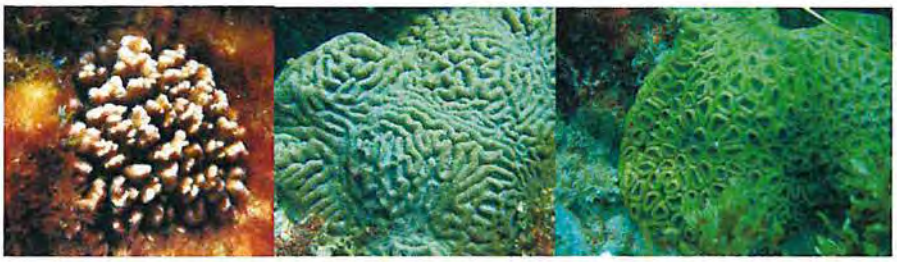
Actiniaria

Ascidiacea



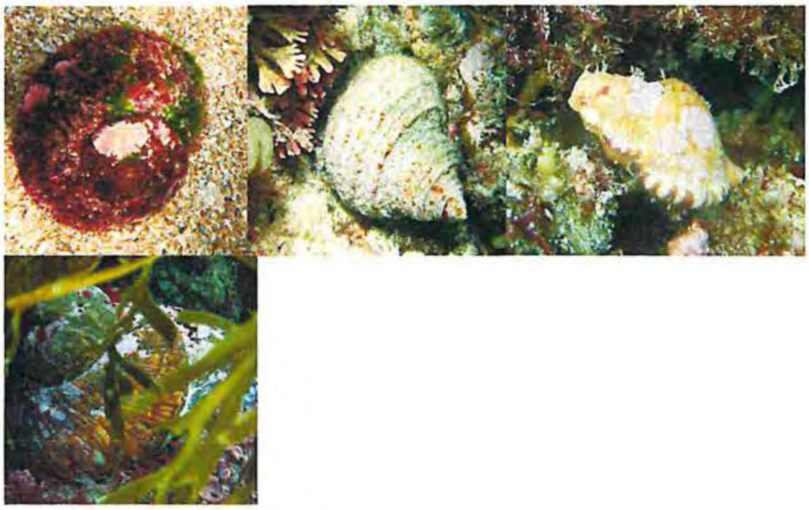
Asteroidea

Scleractinia



Echinoidea

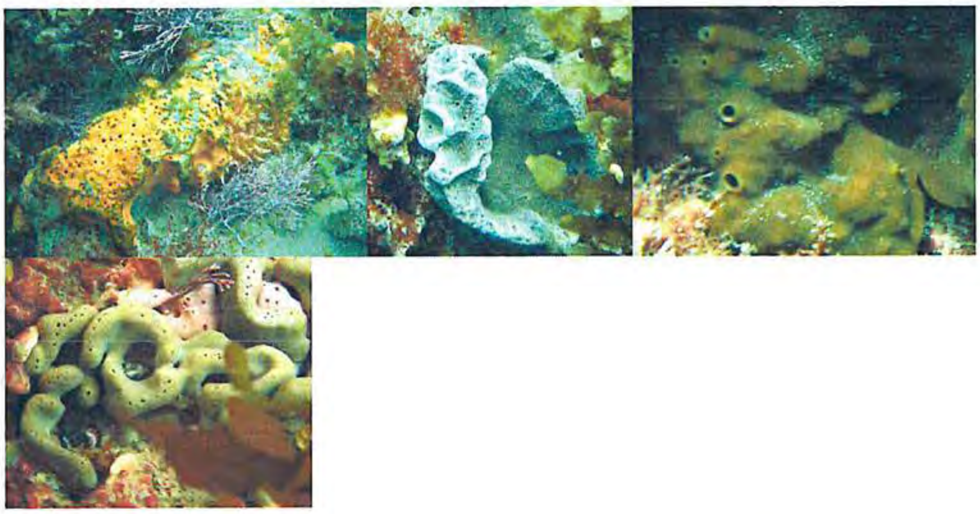
Gastropoda



Holothuroidea

Opisthobranchia

Porifera



Zoanthidea

Appendix 2.

Family, genus, species and common names for each fish species seen in the SBRUV surveys along with distribution (based on Hutchins 1994 and Hutchins & Swainston 1993, **D**- Tropical sp. occur to RI, **E**- Subtropical sp. range Coral Bay – Cape Leeuwin, **F**- Subtropical sp. range Shark Bay – Recherche Archipelago, **G**- Temperate sp. range Abrolhos-Recherche Archipelago, **H**- Temperate sp. range from Shark Bay south and east to eastern states, **I** as H but from Abrolhos, **J**- As H but from RI.) and response to SBRUV categories. Focal species shown in bold with comparative species underlined.

Family	Genus	Species	Common Name	Distribution	Response Group
Aplodactylidae	*Aplodactylus*	*sp*	Western sea carp	Temperate (J)	Non responsive
Apogonidae	*Apogon*	*victoriae*	Red Striped cardinalfish	Sub-tropical (E)	Non responsive
Arripidae	*Arripis*	*georgianus*	Herring	Temperate (I)	Non responsive
Berycidae	*Centroberyx*	*lineatus*	Swallowtail	Temperate (J)	Non responsive
Caesioscorpididae	*Caesioscorpis*	*theagenes*	Fusilier sweep	Temperate (G)	Non responsive
Carangidae	*Elegatis*	*bipinnulata*	Rainbow runner	Tropical (D)	Non responsive
	Pseudocaranx	*sp*	**<u>Silver/Sand trevally</u>**	Temperate (H)	Responsive/ Usually feed
	Seriola	*hippos*	**Samson fish**	Temperate (H)	Inquisitive/ occasionally feed
	Seriola	*lalandi*	**Yellowtail kingfish**	Temperate (H)	Inquisitive/ occasionally feed
	Trachurus	*novaezelandiae*	Yellowtail scad	Temperate (H)	Non responsive
Carcharhinidae	*Carcharhinus*	*Sp*	Shark sp.		Non responsive
Chaetodontidae	*Chelmonops*	*curiosus*	Western talma	Temperate(H)	Non responsive
Cheilodactylidae	*Cheilodactylus*	*gibbosus*	Crested morwong	Sub-tropical (F)	Non responsive
	Cheilodactylus	*rubrolabiatus*	Redlipped morwong	Sub-tropical (F)	Non responsive
	Dactylophora	*nigricans*	Dusky morwong	Temperate (J)	Non responsive
	Nemadactylus	*valenciennesi*	**Queen snapper**	Temperate (J)	Non responsive
Chironemidae	*Threpterius*	*maculosus*	Silver spot	Temperate (I)	Non responsive
Dasyatidae	*Dasyatis*	*brevicaudata*	Smooth stingray	Temperate (H)	Responsive/ Usually feed
Echeneididae	*Echeneis*	*naucrates*	Slender suckerfish	Circum	Non responsive
Enoplosidae	*Enoplosus*	*armatus*	Old wife	Temperate (I)	Non responsive
Gerreidae	*Parequula*	*melbournensis*	Silverbelly	Temperate (J)	Inquisitive/ occasionally feed
Girellidae	*Girella*	*zebra*	Zebrafish	Temperate (I)	Non responsive
Glaucosomatidae	*Glaucosoma*	*hebraicum*	**WA Dhufish**	Sub-tropical (F)	Inquisitive/ occasionally feed
Haemulidae	*Plectrorhinchus*	*flavomaculatus*	Goldspot sweetlip	Tropical (D)	Inquisitive/ occasionally feed
Heterodontidae	*Heterodontus*	*portusjacksoni*	Port Jackson shark	Temperate (I)	Responsive/ Usually feed

Kyphosidae	*Kyphosus*	*cornelli*	Western buffalo bream	Sub-tropical (E)	Non responsive
	Kyphosus	*sydneyanus*	Silver drummer	Temperate (H)	Non responsive
Labridae	*Achoerodus*	*gouldii*	**Blue groper**	Temperate (I)	Non responsive
	Anampses	*geographicus*	Scribbled chieseltooth wrasse	Tropical (D)	Inquisitive/ occasionally feed
	Austrolabrus	*maculatus*	Black spot wrasse	Temperate (H)	Non responsive
	Bodianus	*frenchii*	Western foxfish	Temperate (I)	Inquisitive/ occasionally feed
	Choerodon	*rubescens*	**Baldchin groper**	Sub-tropical (E)	Non responsive
	Coris	*auricularis*	**Western King wrasse**	Sub-tropical (F)	Responsive/ Usually feed
	Eupetrichthys	*angustipes*	Snakeskin wrasse	Temperate (J)	Non responsive
	Halichoeres	*brownfieldi*	Brownfields wrasse	Sub-tropical (F)	Non responsive
	Labroides	*dimidiatus*	Cleaner fish	Tropical (D)	Non responsive
	Notolabrus	*parilus*	**Brownspot wrasse**	Temperate (H)	Responsive/ Usually feed
	Opthalmolepis	*lineolatus*	Maori wrasse	Temperate (I)	Responsive/ Usually feed
	Pictilabrus	*laticlavius*	Senator wrasse	Temperate (I)	Inquisitive/ occasionally feed
	Pictilabrus	*sp*	False senator wrasse	Temperate (G)	Inquisitive/ occasionally feed
	Pseudolabrus	*biserialis*	Red-banded wrasse	Temperate (G)	Responsive/ Usually feed
	Suezichthys	*cyanolaemus*	Bluethroated rainbow wrasse	Sub-tropical (E)	Responsive/ Usually feed
	Thalassoma	*lunare*	Moon wrasse	Tropical (D)	Inquisitive/ occasionally feed
	Thalassoma	*lutescens*	Green moon wrasse	Tropical (D)	Inquisitive/ occasionally feed
	Thalassoma	*septemfasciatus*	Seven-banded wrasse	Sub-tropical (E)	Non responsive
	unknown	*Sp*	Wrasse		Responsive/ Usually feed
Monacanthidae	*Bigener*	*brownii*	Spiny-tailed leatherjacket	Temperate (J)	Non responsive
	Meuschenia	*flavolineata*	Yellow-striped leatherjacket	Temperate (I)	Non responsive
	Meuschenia	*galii*	Blue lined leatherjacket	Temperate (I)	Non responsive
	Meuschenia	*hippocrepis*	Horseshoe leatherjacket	Temperate (I)	Responsive/ Usually feed
	Unknown	*Sp*	Leatherjacket		Non responsive
Moridae	*Lotella*	*rhacinus*	Beardie	Temperate (I)	Non responsive
Mugiloidae	*Parapercis*	*haackei*	Wavy grubfish	Temperate (H)	Non responsive
Mullidae	*Parupeneus*	*signatus*	Blackspot goatfish	Tropical (D)	Non responsive
	Unknown	*Sp*	Goatfish		Non responsive
	Upeneichthys	*vlamingii*	Blue spotted goatfish	Temperate (I)	Non responsive
Muraenidae	*Gymnothorax*	*prasinus*	Green moray	Sub tropical (F)	Inquisitive/ occasionally feed
	Gymnothorax	*woodwardii*	Woodwards reef eel	Sub tropical (E)	Inquisitive/ occasionally feed
Myliobatidae	*Myliobatus*	*australis*	Eagle ray	Temperate (I)	Responsive/ Usually feed

Nemipteridae	*Pentapodus*	*vitta*	Western butterfish	Sub-tropical (E)	Responsive/ Usually feed
Odacidae	*Odax*	*acroptilus*	Rainbow cale	Temperate (I)	Non responsive
	Odax	*cyanomelas*	Herring cale	Temperate (I)	Non responsive
	Siphonognathus	*caninus*	Sharpnosed weed whiting	Temperate (J)	Non responsive
Orectolobidae	*Orectolobus*	*sp*	Wobbegong	Temperate (J)	Responsive/ Usually feed
Ostraciidae	*Anoplocapros*	*lenticularis*	White-barred boxfish	Temperate (I)	Non responsive
	Anoplocapros	*robustus*	Western smooth boxfish	Temperate (H)	Non responsive
Pempheridae	*Pempheris*	*klunzingeri*	Rough bullseye	Temperate (H)	Non responsive
Platycephalidae	*Platycephalus*	*speculator*	Southern blue-spot Flathead	Temperate (H)	Non responsive
Plesiopidae	*Paraplesiops*	*bleekeri*	Western blue devil	Temperate (I)	Non responsive
	Trachinops	*noarlungae*	Yellow headed hulafish	Temperate (I)	Non responsive
Pomacentridae	*Chromis*	*klunzingeri*	Black head puller	Temperate (G)	Inquisitive/ occasionally feed
	Chromis	*westaustralis*	Westralian puller	Sub-tropical (E)	Inquisitive/ occasionally feed
	Parma	*bicolor*	Bicolor scalyfin	Temperate (G)	Non responsive
	Parma	*mccullochi*	McCullochs scalyfin	Temperate (G)	Non responsive
	Parma	*victoriae*	Victorian scalyfin	Temperate (I)	Non responsive
Pseudochromidae	*Labracinus*	*lineata*	Lined dottyback	Tropical (C)	Non responsive
Rhinobatidae	*Trygonorhina*	*fasciata*	Fiddler ray	Temperate (I)	Responsive/ Usually feed
Scorpaenidae	*Neosebastes*	*nigropunctatus*	Black spotted gurnard fish	Temperate (I)	Non responsive
Scorpididae	*Neatypus*	*obliquus*	Footballer rweep	Sub-tropical (F)	Responsive/ Usually feed
	Scorpis	*aequipinnis*	Sea rweep	Temperate (J)	Inquisitive/ occasionally feed
	Scorpis	*georgianus*	Banded rweep	Temperate (I)	Inquisitive/ occasionally feed
Scyliorhinidae	*Aulohalaelurus*	*labiosus*	Black spot catshark	Temperate (G)	Inquisitive/ occasionally feed
Serranidae	*Acanthistius*	*serratus*	Western wirrah	Temperate (G)	Inquisitive/ occasionally feed
	Epinephelides	*armatus*	**Breaksea cod**	Sub-tropical (F)	Inquisitive/ occasionally feed
	Hypoplectrodes	*nigrorubrum*	Black banded seaperch	Temperate (I)	Inquisitive/ occasionally feed
	Othos	*dentex*	**Harlequin fish**	Temperate (I)	Inquisitive/ occasionally feed
Sillaginidae	*Sillaginodes*	*punctata*	**King George whiting**	Temperate (I)	Inquisitive/ occasionally feed
	Sillago	*sp*	**Whiting**		Inquisitive/ occasionally feed
Sparidae	*Pagrus*	*auratus*	**Pink snapper**	Temperate (H)	Responsive/ Usually feed
	Rhabdosargus	*sarba*	Tarwhine	Sub-tropical (E)	Inquisitive/ occasionally feed
Sphyraenidae	*Sphyraena*	*novaehollandiae*	Snook	Temperate (J)	Non responsive
	Sphyraena	*obtusata*	Striped seapike	Tropical (D)	Non responsive

Tetraodontidae	*Torguigener*	*pleurogramma*	Striped toadfish	Temperate (H)	Responsive/ Usually feed
	Unknown	*Sp*	Toadfish		Responsive/ Usually feed
Triakidae	*Furgaleus*	*macki*	Whiskery shark	Temperate (I)	Non responsive
Triglidae	*Unknown*	*Sp*	Gurnard		Non responsive
Urolophidae	*Trygonoptera*	*ovalis*	Striped Stingaree	Temperate (I)	Inquisitive/ occasionally feed