Artificial Reefs: types, applications, trends in deployment and the development of a cost-effective method for monitoring their fish faunas

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This thesis is presented for the Degree of Honours in Marine Science

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Declaration

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Thomas Andrew Bateman

2nd November 2015
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THESIS TITLE: Artificial Reefs: types, applications, trends in deployment and the development of a cost-effective method for monitoring their fish faunas

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Abstract

The focus of this thesis is on the design and use of artificial reefs and the development of a cost-effective method for monitoring their fish faunas. A review of habitat enhancement structures around the world, focusing primarily on artificial reefs, found that these structures have been used for a wide range of purposes such as sediment stabilization, mitigation of illegal trawling, enhancing recreational fisheries and the provision of additional habitat and nurseries for threatened fish stocks. Over time, there has been a growing trend in the use of purpose built reef modules as opposed to the use of materials of opportunity. Within Australia this has been most evident in the shift away from the use of tyres and steel vessels, to the use of specially designed concrete reef modules. As these structures can require financial investments within the millions, it is important to evaluate their effectiveness through post deployment monitoring. A central part of the citizen science monitoring project being developed by Recfishwest in Western Australia is the use of university students to extract information from the Baited Remote Underwater Video (BRUV) footage collected by recreational fishers. This study found that whilst observers recorded similar numbers of species and abundance (total MaxN), significant differences were present between observers in terms of their faunal compositions. This indicates that if inexperienced observers are used in the future as part of a cost-effective monitoring project, observer bias may be a potential source of error in the data and should be mitigated through observer training. Statistical analysis of footage collected from the Bunbury and Dunsborough artificial reefs using BRUVs found a significant difference in species composition between the footage from the two reefs but not between camera positions. However, increased camera soak time and footage collection over a greater temporal scale are needed to increase the reliability of the data. Whilst improvements to the sampling regime are recommended, the use of cost-effective BRUVs shows potential as an effective method for monitoring the fish fauna of artificial reefs using citizen science.
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Chapter 1: General introduction

This chapter provides an introduction to this thesis. It focuses on the rationale of the research undertaken, details how this supports and enhances the research activities of two larger projects on artificial reefs and describes how the thesis is structured.

1.1 Background

Habitat enhancement structures, which include artificial reefs, are structures purposely placed on the substrate that mimic characteristics of natural structural habitat and concentrate populations of marine flora and fauna (Jensen 2002). The practice of creating artificial reefs has been around for thousands of years and the scale of their use around the world ranges from simple artisanal fisheries, to large-scale commercial operations (White et al. 1990, Nakamae 1991, Grove et al. 1994). Artificial reefs have been deployed to suit a broad range of purposes concerning both fisheries management and environmental protection including coastal defence and sediment stabilization (Harris 2009), prevention of illegal trawling and habitat protection (Ramos-Esplá et al. 2000, Jensen 2002), and the provision of additional habitat and nurseries for fish stocks, including threatened species (Pickering et al. 1999, Claudet and Pelletier 2004).

1.2 Relationship to other projects on artificial reefs in Western Australia

This thesis forms part of two broader projects on artificial reefs, as follows.

The first entitled "Can recreational fishers provide a cost effective means for monitoring artificial reefs?" is funded by Recfishwest. This project aims to develop a relationship between recreational fishers and fishery scientists and investigate the potential for a community based, cost effective monitoring program to assess the success of habitat enhancement schemes in attracting target faunal species. Researchers from Murdoch University are the principal investigators on this project.

The second, entitled "The application, needs, costs and benefits of habitat
enhancement structures in Western Australia and cost effective monitoring methods” is funded by the Fisheries Research and Development Corporation (FRDC). Two of the stated aims of that large multidisciplinary project are relevant to this thesis, namely:

1. Identify what habitat enhancement structures are currently available throughout the world and what benefits each type may have for recreational and commercial fishing as well as identifying the benefits for aquaculture and the environment.

2. Determine cost effective methods to monitor habitat enhancement structure developments using easily available materials and data collection by community and industry groups.

Recfishwest is the principal investigator, with Murdoch University, the Department of Fisheries and Ecotone Consulting being project partners. It should be noted at the outset that the role of the research undertaken in this thesis was not to achieve the above-stated aims of these two projects, but to contribute to achieving those goals.

In light of the aims of the above research projects, the contents of this thesis can be divided into two main components; (1) literature-based research on the types of habitat enhancement structure used throughout the world, focusing on the trends in artificial reef construction and their use within Australia, and (2) research and evaluate a cost-effective method for monitoring the fish fauna of artificial reefs. The following two sections outline the background and rationale for each section.

1.3 Habitat enhancement structures

Although there has been growing interest from many government and community organisations in the construction and deployment of habitat enhancement structures for a wide variety of purposes, limited guidelines are available that provide advice on which type of structure(s) are best suited to meet the desired objective. Without such information, there is increased risk of duplicating years of trial and error through suboptimal reef design and ineffective management and incurring large expenses in the process (Diplock 2010). Thus, the completion of a review of the types of habitat enhancement structure available, their design and construction, benefits and drawbacks
and suitability to meeting the aims of the various user groups is required. Given the recent increase in the construction and deployment of habitat enhancement structures in Australia, a critical synthesis of this information could be used to help guide the future development and use of such structures.

When providing guidelines for future deployment of habitat enhancement structures in Australia it is important to reflect on the past use of such structures and investigate how trends have changed as our knowledge in this area has expanded. Such a review of historical trends is possible in Australia as habitat enhancement structures, predominately artificial reefs, have been employed for ~50 years. Although reviews of the types of reefs deployed within Australia have been conducted before (Pollard and Matthews 1985, Kerr 1992, Branden et al. 1994, Coutin 2001), the most recent was undertaken 15 years ago, during which time numerous new habitat enhancement structures have been constructed. Moreover, these early reviews demonstrated that most reefs had been constructed primarily using waste material such as used tyres and decommissioned vessels, whereas recently deployed reefs, such as those in Geographe Bay in Western Australia, have focused on the use of purpose built modules specially designed for the aims of the user groups.

1.4 Monitoring fish communities on artificial reefs

The deployment of purpose-built artificial reefs requires a significant financial investment and it is thus important to determine whether these reefs have achieved their desired goals. In the case of reefs that have been deployed to attract particular fish species (usually those targeted by recreational fishers), it is important to regularly monitor the fish assemblages of these reefs and how these assemblages change over space and time (Carr and Hixon 1997, Pickering and Whitmarsh 1997, Pickering et al. 1999, Holmes et al. 2013). A range of methods have been developed that are available to monitor the fish communities of artificial reefs and other habitats, each of which has its own particular sets of strengths and weakness (see Kingsford and Battershill 1998). One of these methods, Baited Remote Underwater Video (BRUV) monitoring, has
become increasingly popular in recent years, driven by advances in camera and computer technology and a growing demand for the use of non-destructive/extractive monitoring techniques (Harvey and Cappo 2000, Cappo et al. 2003).

Baited Remote Underwater Videos have been widely used to assess fish assemblages in the past and have been found suitable for use in citizen science projects as they remove the need for skilled observers in the field (Langlois et al. 2010, Lowry et al. 2012, Holmes et al. 2013). The use of BRUVs has also been shown to attract a greater number and diversity of fish species, providing a more accurate representation of the whole community than that recorded using unbaited cameras (Cappo et al. 2003, Watson et al. 2010).

Some of the limitations with the use of BRUVs are that the effects of the bait can vary significantly depending on the environmental conditions during deployment, the and to avoid repeated counts of the same fish, only a relative abundance is obtainable (Willis and Babcock 2000, Lowry et al. 2012, Harvey et al. 2013). It has also been noted that there is potential for bias towards predatory species, however, this may be beneficial when monitoring recreationally targeted species as the majority of these species are predatory (Willis et al. 2000, Malcolm et al. 2007). The post field video processing time is also considered a limitation of using BRUVs. Compared to traditional diver underwater visual census, which requires very little analysis post fieldwork, underwater video footage requires detailed lab analysis to extract numerical data (Harvey et al. 2013).

During 2013, two purpose-built artificial reefs were deployed in Geographe Bay, Western Australia, as part of an artificial reef trial project funded by Royalties for Regions ($1,860,000) and from revenue generated from recreational fishing and boat licences ($500,000; Department of Fisheries Western Australia 2015). The aim of the deployment was to attract key recreational fish species such as Pink Snapper (*Chrysophrys auratus*), Samson Fish (*Seriola hippos*) and Silver Trevally (*Pseudocaranx dentex*) and thus enhance local fishing (Department of Fisheries Western Australia 2015). More detailed information on these reefs is provided in Section 4.2.
Given the purpose of these structures, both Recfishwest and the Department of Fisheries Western Australia are involved in the monitoring of the fish fauna of the two artificial reefs. Due to the relatively high cost of scientific monitoring, Recfishwest are particularly interested in the development of more cost-effective monitoring methods using citizen science. Such a monitoring regime would also have the added benefit of engaging local recreational fishers and promoting the reefs in the surrounding areas. Recfishwest has decided to investigate the possibility of employing recreational fishers to carry out long-term monitoring of the Geographe Bay artificial reefs using BRUV systems constructed from low cost materials.

A central part of the artificial reef monitoring approach envisaged by Recfishwest is that university students, as a part of their studies in a relevant area (e.g. marine science), will analyze the footage and extract data on the fish communities of the artificial reefs from the BRUV footage collected by the recreational fishers. However, the use of observers with limited experience in logging data from underwater footage must be accounted for and biases such as that between multiple observers understood and managed. For example, previous observational studies of fish assemblages have found that the scale of difference found between counts and species identification from novice and highly experienced observers was comparable to ecologically meaningful variation if such data represented real differences among sites (Williams et al. 2006). Furthermore, such variation between observers was found to be particularly prevalent when dealing with cryptic and fast moving species (Thresher and Gunn 1986, Thompson and Mapstone 1998, Williams et al. 2006).

Thus the presence of significant amounts of observer bias could clearly compromise the monitoring data set. There is, therefore, a need to investigate the amount of observer bias that might be present among observers with broadly equivalent educational and recreational fishing experience, but limited experience in extracting data from BRUV footage.
1.5 Thesis structure

This thesis comprises six chapters.

Chapter 1. General introduction.
This chapter provides an introduction to this thesis and describes the rationale for undertaking the various research components.

Chapter 2. The design and application of habitat enhancement structures
This literature review summarises information on the types of habitat enhancement structures employed throughout the world. It critically reviews the effectiveness and drawbacks of various designs and construction materials and provides an easy to use pictorial summary (i.e. a heat map) to aid end users in choosing the most appropriate construction material for new habitat enhancement structure deployments.

Chapter 3. Trends in artificial reef construction, design and management in Australia.
This chapter builds on the work of Kerr (1992) and provides a historical overview of how the trends in the characteristics of artificial reefs within Australia (i.e. construction materials, design, location and purpose) have changed over the past 50 years, from the deployment of the first artificial reef in 1965 to the present day.

Chapter 4. Observer bias in the analysis of baited remote underwater video footage.
This chapter determines the level of bias present between four observers with similar educational qualifications and recreational fishing experience, who all viewed the same suite of underwater video footage recorded on an artificial reef using baited remote underwater video.
Chapter 5. Analysis of a cost-effective method for monitoring artificial reefs.
This chapter details the results of an investigation to determine the types of information that can be extracted from baited remote underwater video footage collected by Recfishwest and Ecotone Consulting on two artificial reefs.

Chapter 6. General conclusion.
This chapter provides a brief overview of the key findings of the research undertaken in this thesis and provides some direction for future research.
Chapter 2: The design and application of Habitat Enhancement Structures in the marine environment

2.1 Introduction
Habitat Enhancement Structures (HESs) have been used worldwide for a variety of purposes concerning fisheries enhancement, environmental management and sustainability (Seaman and Spraque 1991, Seaman and Tsukamoto 2008, Bortone et al. 2011). These structures are regarded as “any purpose-built structure or material placed in the aquatic (oceanic, estuarine, river or lake) environment for the purpose of creating, restoring or enhancing a habitat for fish, fishing, and recreational activities” (Department of Fisheries Western Australia 2012). The primary application of HESs in the past has been the enhancement of local fisheries with the most common form of this technology being Artificial Reefs (AR) (Seaman and Spraque 1991, Seaman and Tsukamoto 2008, Bortone et al. 2011). More recent applications of this technology, however, have shown that HESs can fill a variety of roles in, for example, species conservation (Pickering et al. 1999, Claudet and Pelletier 2004), the provision of additional specific types of habitat (Spanier and Almog-Shtayer 1992), aquaculture and sea ranching (Nakamae 1991, Grove et al. 1994, Fabi and Fiorentini 1996), tourism (Branden et al. 1994), illegal fishing mitigation (Ramos-Esplá et al. 2000), habitat restoration (Clark and Edwards 1994), and habitat protection (Jensen 2002).

The design of HESs incorporates both engineering and biological elements. Successful HESs must be designed to meet the needs of their intended purposes (such as enhancing fish stocks) along with the regulatory requirement of structural stability and integrity (Harris 1995). An adequate understanding of the waves and currents at the proposed location as well as the possible changes in the hydrodynamics of the area is essential to developing an effective HES (London Convention and Protocol/UNEP 2009). In addition, the material(s) chosen for the construction of a structure must be appropriate for the intended purpose and their properties thoroughly evaluated and
understood. Although there is great interest in the development and deployment of HESs from many government and community organisations, there are limited guidelines available for the best application of various types of structure. Thus, the aim of this review is to summarise information on the types and uses of HESs around the world.

2.2 History of HES design

The earliest HESs were made from natural, locally abundant materials, such as rocks, logs and bamboo, referred to as “Materials of Opportunity” or MOP (Harris 1995, Harris et al. 1996). In more recent years, however, these natural materials have been supplemented by the use of more modern MOP, e.g. retired ships, car tyres, abandoned oil and gas rigs and concrete rubble (Seaman and Tsukamoto 2008).

The initial frequent use of cheap and abundant materials was due to the fact that the construction of HESs has traditionally been sparsely funded, and that the principal benefactors have been proponents of recreational and commercial fishing and/or scuba-divers (Harris et al. 1996). As a result, many of the ARs constructed in the early 1990’s were poorly managed and have been described as a hit-or-miss dumping operation of unsightly scrap material (Pickering and Whitmarsh 1997). Consequently, much effort has been placed in the design and management of HESs and thus there is growing trend towards the use of purpose-built structures (Pickering and Whitmarsh 1997).

Experimentation with purpose built concrete modules began in Japan in 1952 and was soon followed by the formation of the first national HES program (Grove et al. 1994). The subsequent development of Japan’s Coastal Fishery Enhancement and Development Program (ENSEI) in 1977 provided greater funding and dedicated research into the effects and benefits of new AR materials and designs (Nakamae 1991, Grove et al. 1994, Jensen 2002, Bortone et al. 2011). International collaboration at events such as the International Conference on Artificial Reefs and Related Artificial Habitats (CARAH) has also been a fundamental step in allowing HES researchers from
across the world to share their knowledge on the role that HESs can play in management of fisheries and the marine environment (Grove et al. 1994).

2.3 Materials of Opportunity

Materials of Opportunity used to construct HESs include a wide range of natural materials, *e.g.* rock, shell, or trees and modern human-made materials, *e.g.* concrete debris, retired ships, car tyres, and decommissioned oil and gas rigs. As mentioned previously, due to the low costs, MOP have historically been the dominant material used to construct HESs. Moreover, it is considered as an effective method of recycling the material for productive purposes, whilst simultaneously reducing the cost of constructing HESs (London Convention and Protocol/UNEP 2009). Although modern MOP are generally more durable than their natural counterparts, they also require greater care and management due to the possible negative environmental impacts (Harris 1995, Harris et al. 1996, Seaman and Tsukamoto 2008).

2.3.1. Natural rock and concrete rubble

The use of natural rock to create HESs dates back to the 1600’s, where large rock beds were created on sandy substrates in Japan to increase the harvest of kelp (Nakamae 1991). Nowadays, concrete rubble and natural rock are still some of the most common materials used for constructing HESs and their use as a tool for marine fisheries enhancement has been well documented (*e.g.* Bohnsack and Sutherland 1985, Baine 2001). These materials are employed to provide a hard rocky substrate, which can then be colonised by a wide variety of epifaunal species (Jensen 2002). Thus, this type of AR has shown to be very successful in creating algal beds (particularly for kelp (Nakamae 1991)), providing rock lobster habitat (Pickering and Whitmarsh 1997) and mitigating against the effects of rocky habitat loss (Hueckel et al. 1989).

The popularity of this material is partially due to the fact that it can be easily sourced from a variety of places such as construction sites, demolished buildings and
bridges, local quarries and channel dredging works. Rock and concrete rubble are also often deployed by simply dumping the material off a barge into the sea using bulldozers, reducing both the cost and time required for deployment (Fig. 2; Lukens and Selberg 2004). Comparisons between HESs constructed from rock, prefabricated concrete shelters and steel vessels off the southern California coast found that quarry rock was the preferred reef material even though it was less effective than prefabricated concrete shelters in attracting fish. Moreover, rock was considered a better material than the others due to its low cost, ease of handling, and reduced scouring and sedimentation around the HES (Turner et al. 1969).

Fig. 2.1: Dumping of quarry rock and concrete rubble for the creation of an offshore artificial reef. Reproduced from Nell (2010).

Although, once submerged, both concrete rubble and natural rock have been found to be very durable and stable, some material may contain high levels of heavy metals, which can be liberated into the environment by leaching and the materials should be assessed prior to deployment (London Convention and Protocol/UNEP 2009).
Chapter 2

Rocks with high amounts of quartz are particularly favourable as this mineral is composed of silicon dioxide, one of the main components of many natural reefs and is fully ‘compatible’ with the environment (London Convention and Protocol/UNEP 2009).

Another consideration when using rock or concrete rubble in constructing HESs is the size of the rocks used. For most applications, large rock is preferred as it provides more interstitial space. In contrast, small stones have been found to pack tightly and the resultant spaces may easily be filled with sand, gravel and rock chips (Lukens and Selberg 2004).

2.3.2. Tyre reefs

Each year, millions of tyres are produced throughout the world. Although some are able to be reused through retreading or burnt as fuel, the majority are disposed of in landfill sites (Collins et al. 1995, Collins et al. 2002). As a result, used tyres have been widely used in the construction of HESs. However, they are now regarded as unsuitable for use in the marine environment following a number of poorly designed and managed projects. The initial view was that once submerged underwater, tyres would be protected from ultraviolet degradation and the stable chemical environment would help limit the leaching of chemicals from the rubber (Collins et al. 2002). The open shape of the tyre, which causes problems for land disposal, was seen as an advantage when creating HESs because it creates multiple habitats, and thus niches and shelter for juvenile fish and invertebrates (Collins et al. 1995). In light of this, the use of used tyres in AR construction appeared an excellent solution by recycling tyres in an ecologically amenable way and proving cheap and readily available materials for the creation of additional fish habitat (Sherman and Spieler 2006).

A review by Collins et al. (1995) found that ARs constructed from tyres were most abundant in south-west Pacific and the Atlantic seaboard of the USA. For example, > 70 tyre reefs have been built along the Atlantic seaboard and 54 tyre reefs
deployed in Malaysia using > 1.5 million individual tyres (Kerr 1992). This material has been widely used in Australia too, with over 30 tyre reefs having been constructed.

Tyre ARs, when initially deployed, were found to be effective at attracting target fish species, particularly in Australia where valuable recreational and commercial species such as *Sillaginodes punctatus* (King George Whiting) and *Chrysophrys auratus* (Pink Snapper) were caught in higher numbers around these reefs than around nearby natural reefs (Branden et al. 1994). Soon after deployment, however, monitoring revealed that the positive buoyancy of the tyres was causing many of the reefs to break apart and wash up on nearby beaches (Collins et al. 1995). Another major concern with the use of tyres is their tendency to flex during storms, causing any rigid epifaunal species on the tyres surface to be dislodged and thus ‘lost’. This finding is exemplified by the study of Fitzhardinge and Bailey-Brock (1989) who compared the growth of corals on tyres, concrete and metal in Hawaii and observed that the latter two substrates were far more effective at increasing coral biomass due primarily to the flexing of the tyres in rough weather.

The Osborne tyre reef in Florida, USA, is a clear example of how poorly designed HESs can cause more harm than good to the surrounding environment. In 1967, local fishers and environmental resource managers initiated a project to build a large tyre AR offshore of Broward County. The reef was built using 1-2 million banded, but un-ballasted, car and truck tyres (Sherman and Spieler 2006). Since deployment of that reef, storms and strong ocean currents have caused the bands to give-way and break apart the reef, causing many of the tyres to wash up on nearby beaches. Those that remain in the water can still be identified by brand name owing to a complete lack of epifaunal growth (Fig. 2.2). Moreover, many continue to drift along the benthos causing severe damage to nearby natural reefs (Sherman and Spieler 2006).
Some studies have demonstrated that heavy metals such as zinc, which represent 1-2% of the weight of a tyre, leach and accumulate on hydroids (*Halecium* spp.) growing on the tyre’s surfaces (Collins et al. 1994). These findings, combined with the previous experience of tyre ARs, have led to the use of tyres in the marine environment being discouraged and even banned in a number of countries. Moreover, such is the level of concern; several environmental groups have recommended the removal of existing tyre reefs (Dorer 1978, Sherman and Spieler 2006).

### 2.3.3. Retired ships and other steel-hulled vessels

The long history of accidental shipwrecks on the seafloor has allowed the value of ships as HESs to be well studied over many years (London Convention and Protocol/UNEP 2009). Steel-hulled vessels, selected for their hull integrity, are considered by many AR builders to be a durable material in the marine environment. For example, such structures may last for > 60 years, depending on vessel type, physical condition, location of deployment and the severity of local storms and wave action (Lukens and Selberg 2004). A study of the disposal and recycling options available for retired ships
carried out by the United States Navy determined that deploying those ships as ARs provided the best economic outcome. This is because, as well as being the least expensive disposal method, the reefing of retired ships has the potential to provide economic offsets, such as increased revenues from tourism, recreational diving, sport fishing and improved commercial fishing (Hess et al. 2001).

Steel vessels generally have a high vertical profile and large surface area for colonization by epibenthic species, which makes them effective at attracting both pelagic and benthic fish species (Lukens and Selberg 2004). Although primarily used for tourism and sport fishing purposes, the reefing of retired vessels has been suggested for a number of other HES purposes, such as providing deep water nurseries for species under heavy fishing pressure such as *Epinephelus itajara* (Atlantic Grouper) (Hess et al. 2001, Lukens and Selberg 2004).

One of the most successful retired ship projects commenced in Queensland, Australia in the late 1960's. The project scuttled vessels with the aim to increase the diversity and abundance of flora and fauna and to enhance particular fish stocks within the local area. Between 1968 and 1990, 14 vessels were scuttled in waters off Moreton Island at depths of 10-22 m to create the Curtain Artificial Reef. Encouraged by this reef's success, other derelict vessels were acquired over a period of 20 years and strategically placed on the seabed to allow divers to navigate easily from one vessel to another without surfacing (Branden et al. 1994). The Curtain AR has since become a major tourist attraction and supports abundant fish life, with good populations of *Epinephelus lanceolatus* (Queensland Grouper), *E. damelli* (Black Rockcod), *Rachycentron canadus* (Cobia) and *Acanthocybium solandri* (Wahoo), as well as several species of elasmobranch (Branden et al. 1994).

There are a number of considerations, however, when deploying retired ships for use as HESs including the presence of pollutants such as polychlorinated biphenyls, radioactive material, petroleum products, heavy metals (*e.g.* lead, mercury and zinc) and asbestos, which are all commonly present in retired ships and thus need to be removed before the vessel can be sunk (Hess et al. 2001, Lukens and Selberg 2004).
Consideration also has to be given to the fact that steel hulls are not as suitable as rock or concrete for colonization by epibenthic flora and fauna due to the sloughing of steel from corrosion (Gregg et al. 1994).

2.3.4. Rigs to Reef

The structures used as offshore oil and gas production platforms, hereafter collectively referred to as “rigs”, have been deployed as ARs. Such structures have been shown to host large and diverse fish communities (Seaman et al. 1989, Love et al. 1994, Rooker et al. 1997). For this reason, many rigs provide a recreational opportunity for both sport fishing and scuba diving (Stanley and Wilson 1989, Love and Westphal 1990, Schroeder et al. 2000). Rigs-to-Reefs (RTR) is the practice of converting decommissioned offshore rigs into ARs. This program, developed by the Minerals Management Service in the USA, aims to preserve established habitat and productive fishing grounds and reduce the cost of removing decommissioned rigs by deploying them as ARs. Such reefs have already successfully been created from decommissioned rigs in the United States, Brunei and Malaysia (Twomey 2010).

Once an oil or gas structure is properly plugged and abandoned, there are three removal options available for converting the structure into an AR (Dauterive 2000). The first method, which is most common in deeper applications, uses explosives to sever the jacket legs of the rig, causing the structure to topple over to a horizontal position on the sea floor (Fig. 2.3). Although this method offers the lowest costs and time, it has the disadvantage of using explosives, which can harm marine life associated with the structure and it eliminates shallow and mid-water habitats (Klima et al. 1988). However, studies have shown that the portions of the rig damaged during reefing are quickly recolonized by fauna (Schroeder and Love 2004).

The second method employs divers to sever the structure at the base using mechanical or abrasive cutters, followed by the lifting of the entire structure from the seafloor and towing it to a new location (Fig. 2.4). This method is typically only used for rigs in water less than 30 m due to the safety of the divers. Although this method is
expensive and labour intensive, it causes minimal damage to the marine habitat on the rig (Lukens and Selberg 2004).

The third removal method involves the partial removal of the upper portion of the rig, which is then placed on the sea floor next to the standing bottom portion (Fig. 2.5). This method allows the bottom portion of the habitat to stay intact, whilst the top portion provides a lower profile to complement the standing section, and increases the overall surface area of the structure for habitat enhancement relative to toppled methods (Dauterive 2000).

Studies by Stanley and Wilson (1997) on the effects of these three different methods of converting rigs to reefs demonstrated that the density and size of fishes were greater near the surface than the bottom of standing oil and gas platforms and that partially removed platforms had a slightly higher fish density than toppled platforms. The size of the rig, water depth, distance from shore, proximity to final reef site and potential resale value of the rig have been identified as the primary factors dictating whether it is cost effective for an obsolete platform to become a permanent reef (Wilson et al. 1987, Lukens and Selberg 2004).
Fig. 2.3: Toppling method for reefing of decommissioned rigs. Reproduced from Dauterive (2000).

Fig. 2.4: Sever and lift method for reefing of decommissioned rigs. Reproduced from Dauterive (2000).

Fig. 2.5: Partial sever method for reefing of decommissioned rigs. Reproduced from Dauterive (2000).
2.4. Purpose-built Habitat Enhancement Structures

Experimentation with purpose-built reef modules began in 1952, following the development of a government subsidy program in Japan and has since gained popularity worldwide (Grove et al. 1991, Pickering and Whitmarsh 1997). The trend towards the construction and use of purpose-built HESs began when directed efforts were made to take advantage of new knowledge on fish behaviour and oceanic processes (Grove et al. 1991, Grove et al. 1994, Jensen 2002). The merging of knowledge on fish behaviour and the physical environment gave HES designers a more rational approach to developing HESs that could target specific species and environments (Grove et al. 1991).

In contrast to earlier structures, these engineered designs did not incorporate MOP into their construction. Thus the materials utilized to construct purpose-built HESs are able to be selected for their durability, resistance to corrosion/abrasion, strength, structural/design demands and compatibility with the marine environment (Lukens and Selberg 2004). In countries such as Japan and Korea, all new HES modules are required to be tested and monitored for at least two years before government assessment determines whether they can be deployed within public waters (Diplock 2010).

Concrete has been found to be a very favourable material for reef construction following a number of HES trials (Pickering and Whitmarsh 1997). This material has been found to be durable in seawater, easily moulded to different specifications and has a similar epifaunal community development to natural coral reefs (Fitzhardinge and Bailey-Brock 1989). Steel is also a very popular material in AR construction and is often used in combination with concrete and for larger AR structures due to the weight considerations of concrete (Lukens and Selberg 2004).

2.4.1 Fish Aggregating Devices

Association with floating structures in open waters during one or more life history stages has been recorded for > 300 fish species belonging to 96 families (Castro et al.
2002). It is generally believed that fish utilise these objects primarily for protection from predators (Hunter and Mitchell 1968), as a meeting location (Soria et al. 2009), as a source of food and to increase the survival of eggs and juveniles (Gooding and Magnuson 1967). A Fish Aggregating Device (FAD) is any purpose-built, moored and positively buoyant (floating or submersible) structure that is designed to attract and/or aggregate fish in order to facilitate fishing activities (Department of Fisheries Western Australia 2012). Fish Aggregating Devices are generally classified as either surface, mid-water or drifting and can be made from a wide variety of materials. Floating palm fronds with rock or concrete filled tyre anchors are still used extensively in many artisanal fisheries, and although generally short lived (i.e. 3-6 months), they effectively attract large numbers of pelagic fish species such as Coryphaena hippurus (Dolphin fish) and Katsuwonus elemis (Skipjack Tuna) (White et al. 1990, Grove et al. 1991).

Technological advances and the development and use of more durable FAD designs and materials have made modern FADs an important tool in many commercial and recreational fisheries. Modern anchored FADs can be placed in waters up to 2000 m deep and incorporate flashers and netting to attract fish and increase structural complexity, respectively (Gates et al. 1996). These devices are commonly employed by recreational fishers and may reduce the fishing pressure on demersal species, by making it easier for fishers to target faster growing and more abundant pelagic species, which aggregate at the FADs (Dempster and Taquet 2004). Drifting FADs, on the other hand, are primarily used by commercial purse seine fishermen to congregate large schools of tuna (Bromhead et al. 2003). To this end, GPS locaters and fish sonars are often incorporated into their design allowing them to be easily tracked and allowing fishers to detect the school size and even fish species congregating at the FADs (Castro et al. 2002).

Although beneficial to some areas by reducing fishing pressure on benthic species, the over use of FADs can lead to depletions in pelagic fish stocks. Thousands of FADs are deployed each year and overall global information on their use is limited (Macfadyen et al. 2009). As well as this, poorly constructed FADs have a tendency to
break away from their mooring during heavy seas and may become a serious navigation hazard.

2.4.2 Benthic Production Reefs

The first generation of purpose-built Benthic Production Reefs (BPR) was developed in Japan in the 1950’s to enhance those local fisheries that were depleting (Bohnsack 1987a). Initially these reefs had simple designs that consisted of small, hollow concrete cubes or cylinders with “windows” in the sides. However, due to their success, by the 1990’s there were over 100 different designs in use, each developed for different species and environmental conditions (Grove et al. 1994). The success of these BPRs showed that despite the greater initial cost involved, the use of reef designs that incorporated not only the biological requirements of target species, but also the engineering aspects relating to material design, placement and performance, produced far greater benefits than earlier reefs made from MOP (Pickering and Whitmarsh 1997).

Benthic Production Reefs are designed not only with the aim of attracting and congregating fish, but also to permanently create more productive fishing areas through the creation of additional habitat and nursery grounds. The ability to deflect horizontal ocean currents upwards, thereby inducing upwelling, has also been incorporated into the design of some BPRs (Bohnsack and Sutherland 1985, Grove et al. 1994). Many of the world's most productive fishing grounds occur in regions where, to compensate for the offshore migration of the surface water, bottom water wells up toward the shoreline, bringing with it nutritious seston and the primary organisms thriving in them (Grove et al. 1994).

2.4.2.1. Design of benthic artificial reefs

Size, shape, void space and the number and size of openings are all important factors in the design of BPRs and reef requirements vary greatly depending on the target species and environmental conditions (Pickering and Whitmarsh 1997). Studies done in Korean waters found marked preferences among different species for particular reef designs and
a significant relationship between reef structure and catch volume (Kim et al. 1994, Lee and Kang 1994). Dice shaped reefs were found to be the preferred habitat of rockfish, whilst turtle dome reef units attracted primarily demersal species (Lee and Kang 1994). Cylinders with large holes along the sides were found to be effective at attracting finfish, with larger, hollow structures consistently found to have the highest species diversity (Kim et al. 1994). Other studies have also shown that whilst larger individual reef units have been found to hold greater biomass densities, these populations are generally made up of larger but fewer individuals. Multiple smaller reefs, on the other hand, were shown to attract greater numbers of smaller individuals and species and are recommended in preference to a single larger reef unit in terms of overall recruitment (Bohnsack et al. 1994). It has been noted, however, that smaller reef units have limited value as nurseries for juvenile fish and for increasing overall production, and thus, in terms of enhancing fisheries, larger reef units with a combination of unit sizes is most effective (Moffitt et al. 1989, Bohnsack et al. 1994).

Direct relationships have been identified between increased reef production and reef volume up to a critical point of 4000 m$^3$, with reef areas between ~2300 m$^2$ – 4600 m$^2$ required to reach equilibrium and permit propagation (Ogawa et al. 1977, Bohnsack et al. 1994). Reef height will also greatly influence the species composition at the reef, with taller individual reef modules being more effective at attracting transient pelagic species. Demersal and benthic species however such as lobsters, which rarely venture above 1m from the seabed will be more affected by the horizontal spread of the reef rather than the vertical height (Bohnsack 1987b).

The structural complexity, particularly the presence and variety of crevices, also plays a significant part in species composition and productivity of an AR. Topographically complex ARs, in comparison with more simplistic shapes, are found to have greater numbers and species of fish associated with them due to the greater number of individuals able to find shelter from predation (Clark and Edwards 1994). Another important consideration in AR design is that fish will generally not venture into dark, closed compartments with only a single exit, preferring spaces with many
openings and a free flow of water. For small fish in particular, which require places to rest, the deployment of AR units at right angles to strong currents can provide effective shelter on the lee side (Dean 1983).

2.4.2.2. Shallow-water benthic artificial reefs

Shallow-water (10-30 m) benthic AR structures are generally constructed from concrete as mentioned previously, due to its moldability and structural integrity in the marine environment (Fitzhardinge and Bailey-Brock 1989, Pickering and Whitmarsh 1997). The “Fishbox” design, which has been used on the east and west coasts of Australia, is a 17 ton hollow concrete cube unit with a reinforced concrete cross brace (Fig. 6; HaeJoo 2015b). These structures were designed to attract a wide range of recreationally important fish species by creating complex spaces and habitats and diverting nutrient-rich water up the water column (Department of Fisheries Western Australia 2015). After only two years of deployment the reefs have seen the number of fish species in the vicinity of the reefs quadruple, with a high presence of target species such as *Chrysophrys auratus* (Pink snapper) and *Pseudocaranx dentex* (Silver Trevally) (Department of Fisheries Western Australia 2010, 2015).
Benthic Production Reefs have also been successfully used in the aquaculture of molluscs, where often the environmental conditions are ideal but there lacks hard substrate within the vicinity for the organisms to attach themselves and establish a population (Badalamenti et al. 2002, Fisheries Research and Development Corporation 2015). A number of ARs have been trialled for use in the farming of wild abalone, a highly valued and sought after product, in Flinders Bay, Western Australia (Fisheries Research and Development Corporation 2015). Some of the benthic reef types trailed include solid concrete blocks, hollow concrete blocks, round concrete tubes and standard besser blocks. The most recent and successful reef design is a purpose built module designed specifically for abalone ranching. Each unit is 1.2 m² by 600 mm high, with a total surface area of 4.5 m² (Fig. 7; HaeJoo 2015a). The design includes improved shelter for juveniles to increase survival rates and grooves in the concrete to help trap waterborne macro algae and improve the food supply. Each reef unit is predicted to harvest up to 10 kilograms of abalone annually, equating to roughly 30 individuals (Fisheries Research and Development Corporation 2015).
2.4.2.3. Deep-water benthic artificial reefs

Deep-water (30-150 m) benthic ARs are generally larger than shallow water ARs and are constructed from steel, making them lighter and easier to deploy than concrete units of similar size. These reefs are effective at congregating pelagic fish species as well as providing shelter and protection for deep-water demersal species. The “Fish Cave” reef units, which have been deployed at the Wild Banks Reef within Moreton Bay, Queensland, are a good example. These units are 11 m tall, 11 m wide, weight 14.4 ton, and were designed to increase the numbers of pelagic species such as mackerel and wahoo which are targeted by recreational fishers and spearfishermen (Queensland Department of National Parks 2015). The title of the world's largest reef unit is held by a steel reef named the "Ocean Cross". This reef unit was installed in 1991 in the Sea of Japan and has a bulk volume of 3600 m³, with a height of 9 m and maximum horizontal dimension of 27 m (Grove et al. 1994).

Fig. 2.7: An abalone reef module in Flinders Bay, Western Australia. Reproduced from HaeJoo (2015a).
2.4.3. Habitat protection reefs

Anti-trawling reefs are generally constructed from a combination of concrete blocks and steel arms that aim to catch and tear through trawling nets. Although these structures are generally less complex and provide less diverse habitat than other purpose built ARs, their extra weight and profile make them effective deterrents to illegal trawling (London Convention and Protocol/UNEP 2009).

European fisheries have been deploying ARs regularly over the past 30 years with the majority aimed at protecting their Mediterranean seagrass beds, which have been severely damaged in the past due to illegal trawling (Jensen 2002). Following baseline surveys in the waters off the SE Iberian peninsula which showed that up to 48% of the Posidonia oceanica meadows had been damaged by trawls, 1.5 m³ concrete blocks with 0.5 m steel arms were deployed over an area 5 400 000 m² to help deter illegal trawling in the marine reserve (Sánchez-Lizaso et al. 1990).

Since the deployment of 358 blocks in 1992, no illegal trawling has been recorded in the area (Ramos-Esplá et al. 2000). Post deployment monitoring of the damaged seagrass beds in the Tabarca reserve has shown promising results with P. oceanica shoot density increasing from 10 to 60 shoots per m² in the 6 years after deployment of the reef (Jensen 2002).

2.4.4. Electrodeposition reefs

Electrodeposition is the process of accreting calcium and magnesium salts on to a cathode using a low electric current. This process was first described by Hilbertz in 1977, and can be used to grow extremely hard calcium carbonate limestone deposits on any steel template (Hilbertz et al. 1977). The “Biorock” material is primarily made up of the mineral aragonite, the same compound that makes up coral skeletons, making it ideal for use in the marine environment (Goreau 2012). Initial studies with this technology found that within 2 months, a coating with a thickness of 5-10 mm of material can be formed around the template (Van Treeck and Schuhmacher 1999).

The process has found use all across the Caribbean, Pacific, Indian Ocean, and Southeast Asia, with most projects taking place in Indonesia (Goreau 2014). As the
template used to grow the material can be made into any shape, it can be utilized for a number of purposes such as repairing damaged coral reefs and the creation of custom dive sites. These effects however are not residual and only occur only when the electrical field is on. As the technology is still experimental, there is also limited research into the possible harmful chemicals that may be produced as by-products of the reaction (Lukens and Selberg 2004).

2.4.5. Artificial seagrass

Seagrass meadows are extremely valuable coastal ecosystems, both ecologically and economically. They provide a number of high-value ecosystem services and it has been proven that fisheries revenue from an area increases when the size and condition of its seagrass meadows improve (Orth et al. 2006, Shahbudin et al. 2011). Additionally, seagrass meadows provide effective sediment stabilization and reduce wave energy, thus providing significant coastal protection (Orth et al. 2006). The reduced hydrodynamic conditions and stabilised sediment caused by the meadows create conditions more suitable for the seagrass itself, enabling further meadow growth in a positive feedback loop (Shahbudin et al. 2011). Unfortunately, seagrass meadows around the world are declining for various reasons with reported cases of seagrass loss increasing tenfold over the last 40 years in both tropical and temperate regions (Orth et al. 2006).

Artificial seagrass has been used extensively in seagrass community research, as the artificial beds can be placed next to natural meadows and easily sampled without damaging the natural seagrass (Virmstein and Curran 1986, Bartholomew 2002). Artificial seagrass has also been widely used as a soft engineering method to protect shorelines from erosion and as an alternative habitat for various marine organisms (Shahbudin et al. 2011). Artificial seagrass beds can be made from a range of materials and customized to mimic the target seagrass species. Studies on artificial seagrass made from green polypropylene ribbons designed to mimic *Thalassia testudinum* (Turtle
grass) showed extremely rapid colonization by seagrass-associated epifauna (Virnstein and Curran 1986).

A major consideration when using artificial seagrass is that the material used in their construction is generally not biodegradable and rough weather may cause the beds to break apart, leading to possible negative environmental impacts. Future research in seagrass rehabilitation includes the development of biodegradable artificial seagrass that can be placed in locations that are generally suitable for seagrass re-establishment, but are sub-optimal for initial settlement. This artificial meadow would then provide the ecosystem engineering function to promote the growth of natural seagrass in its vicinity and then simply biodegrade, thus reducing any disturbance by removing the artificial beds (Innorex 2014).

2.4.6. Multi-Function Artificial Reefs

Multi-Function Artificial Reefs (MFARs) are offshore, underwater structures that can be designed to protect coastlines, reduce erosion, enhance marine habitats and provide a valuable recreational resource. The addition of environmental and recreational amenity to coastal protection facilities provides a range of benefits, which come at a critical time when shoreline modification is accelerating (Black 2001). Additionally, offshore coastal protection does not impair visual amenity and can mitigate the need for rock emplacements along the shoreline that can isolate people from the coast (Harris 2009).

Multi-Function Artificial Reefs will generally have a primary goal such as reducing beach erosion, and a variety of secondary goals such as providing recreational dive sites and increasing local marine biomass. One example of this is the use of AR modules for constructing submerged breakwaters for shoreline stabilization. Breakwaters work by causing larger waves to break on the structure whilst allowing smaller waves to pass unaffected. This allows normal coastal processes to occur in the lee of the reef, whilst effectively reducing the wave energy of larger waves and stabilizing the adjacent beach (Harris 2009).
Along the Southern Caribbean shore of the Dominican Republic 450 Reef Ball units were installed to form a submerged breakwater for shoreline stabilization, environmental enhancement and eco-tourism. The units used for the breakwater were 1.2m high Reef Ball units and 1.3m high Ultra Reef Ball units, with base diameters of 1.5 and 1.6m, and masses of 1600 to 2000 kilograms. The breakwater was installed in water depths of 1.6 m to 2.0 m, to suit the areas 0.4 m tidal range (Harris 2009). Monitoring over the next three years saw the shoreline increase by over 10m, with no adverse impacts recorded on adjacent beaches. As well as stabilizing the shoreline the reef provided a popular diving and snorkelling site owing to the enhanced marine habitat (Harris 2009). Similar projects have been carried out in other parts of the Caribbean, including the Cayman Islands, where a 5-row submerged breakwater reef has protected the shoreline from two category five hurricanes and still remains stable (Harris 2009).

Artificial Reefs deployed as breakwaters need to be designed to withstand the large forces created by breaking waves, wave induced currents, and scour that occurs in the surf zone. For units placed on hard substrate, the main concerns are the strength of the unit and its resistance to sliding and over turning. The weight of the individual unit will contribute to its overall stability and may require pinning to the seafloor for additional stability. For units placed on sand, scour and settlement are the primary concerns and can be prevented by either drilling rods into the substrate through the unit at an angle or by placing the units on an articulated mat (Harris 2009).

Another type of MFAR is an artificial surfing reef, which provides both social and economic benefits through activities such as surfing, diving, fishing and tourism. Artificial surfing reefs also aid in stabilizing beaches much in the same way as shallow water breakwaters and have been used widely in Australia and New Zealand in place of rock walls or other shoreline defences (Mead and Black 1999, Jackson and Corbett 2007). Although the use of breakwaters for shoreline protection is not new, the recent development of more subtle and versatile offshore coastal protection has become
achievable though sophisticated modelling programs that aid in the design process (Jackson and Corbett 2007, Mead 2009).

An example of one of these reef types is the Narrowneck Artificial Reef on the Gold Coast, Australia, constructed for both coastal protection and improved surfing. Built in 1999-2000, the reef was constructed using over 400 sand-filled geotextile containers that were dropped into place using a hopper dredge (Jackson and Corbett 2007). Although the reef requires long period, clean swell to replicate the modelled waves, it has proven successful in improving the surfing quality in the area and often waves that break on the reef will link up to waves on the inner sand bars, significantly increasing the quality of the surf break (Jackson et al. 2007).

The reef has also provided a suitable substrate for development of a diverse ecosystem and has become a popular fishing and diving location. As a result it has been designated as a no anchoring zone to preserve the current growth. Additionally, the type of geotextile used promotes soft growths that do not present a safety hazard to surfers (Jackson and Corbett 2007). One consideration when using this reef type is the possibility of needing to remove the reef if adverse effects become present. Multi-function artificial reefs should be designed with this in mind and a removal method should be determined before placement.

2.4.7. Urban waterfront habitat enhancement

Recent research into the use of HESs has looked at incorporating this technology into structures along urban waterfronts. As construction along coastlines and within canals continues to increase, especially in countries such as Australia where the vast majority of people live along the coast, alternations to the natural movement of sand, as well as removal of habitat has reduced much of the nearshore biodiversity. Urban structures built in the marine environment are generally not designed or managed to provide habitats for the communities of marine organisms that could colonize them. However, by incorporating the current knowledge of nearshore marine communities and ARs, the
urban waterfront may be capable of supporting a significant proportion of regional aquatic biodiversity (Duffy-Anderson et al. 2003).

Two key differences have been identified between natural rocky shores and human-made structures; slope and microhabitat availability (Dyson 2009). Seawalls and other nearshore infrastructure generally provide vertical habitats, whereas rocky shores have very heterogeneous topography (Chapman 2003, Lam et al. 2009). This limits the type and distribution of many intertidal plant and animal species on human created structures (Bulleri and Chapman 2010). To counter this, researchers have suggested designing and building structures such as seawalls with a combination of surface slopes and textures (Dyson 2009). Incorporating microhabitats such as cavities which retain water during low tide and other analogous features, which are lost when rocky shores are developed, can also have a positive effect on the shoreline biodiversity (Dyson 2009).

Recently built seawalls in Sydney Australia have already incorporated this research into their design with the aim of incorporating intertidal habitats into seawalls in a cost-effective manner that neither compromises safety or engineering requirements (Fig. 2.8). Pools created within the Sydney Harbour seawalls have shown to increase the diversity of species of algae and sessile animals many-fold, especially higher up the shore line where environmental conditions are harshest (Bulleri and Chapman 2010). Additional factors such as surface texture, shading, connectivity and water flow have all been identified as affecting nearshore marine communities. By considering these aspects in the design of future projects, diverse communities, similar to those found on natural rocky shores can be established (Moreira et al. 2006). This would provide not only ecological benefits but also cultural, recreational and educational values to the users of these coastlines (Dyson 2009).
2.5. Application of habitat enhancement structures

A wide variety of designs, configurations and materials exist that can be utilized when developing a HES. Ensuring the material and design used are suited to the purpose will maximize the potential benefits from a HES. Prior to commencing a HES project, factors such as location, intended purpose, target species, cost, government regulations and environmental impact should be carefully considered. The following heat map matches the designs and materials used for creating HESs with their effectiveness for a range of different factors (Fig. 2.9).

This heat map aims to provide a simple guide as to the most suitable HES for specific purposes and environments. It can also be used to show that it may be effective to combine a number of different materials and designs in order to gain the maximum benefits from a habitat enhancement project. For example the high vertical profile of a large steel hulled ship will attract large numbers of pelagic fish whilst placing rock beds nearby will increase the amount of hard substrate for rock lobster and other benthic species to inhabit, making these two materials a good combination when designing a recreational fishing and diving reef. It should be noted, however, that the effectiveness of any of these structures and the materials used in their manufacture depends strongly on their engineering quality and management, which has not been covered in detail here (Collins et al. 1994, Pickering and Whitmarsh 1997).
### Fig 2.9. Heat map of the benefits and cost of deploying different types of HESs in different environments. Note: the cost and legislation requirements have been based on Australian government requirements.

**Key**

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Materials of opportunity
- Used car tires
- Ships
- Rock
- Concrete rubble
- Oil and gas rigs

Purpose built structures
- Small concrete benthic modules (<4m³)
- Large concrete benthic modules (>4m³)
- Steel + concrete combination modules
- Large steel modules (eg. Fishcave)
- Multi function artificial reef
- FAD (artisanal)
- FAD (modern/synthetic)
- Electrodeposition
- Artificial seagrass

Heat map colors:
- Green: Very effective
- Brown: Effective
- Gray: Neutral
- Pink: Negative effect
- Red: Strong negative effect
2.6. Conclusion

Research into the past and present use of HESs has allowed for significant advances in both the management and design of these structures. Modern purpose-built reefs have clear advantages over earlier structures, including; lower negative environmental impacts, increased longevity, and effective species-specific designs. However, the increased cost of purpose-built HESs has meant MOP are still frequently used within artisanal fisheries that struggle with budgetary constraints. In such circumstances, effective management and planning will determine the success of such projects.

As detailed here, HESs can be an important tool in the management of fisheries, capable of producing more productive fishing grounds, protecting threatened habitats and species along with added tourism and recreational benefits. The incorporation of this technology into future urban waterfront infrastructure will also aid in improving the biodiversity of modified coastlines and harbours. The current review should allow project managers to easily identify which HESs will be most suited to their intended purpose and help guide the future development and use of HESs around the world.
Chapter 3: Trends in artificial reef construction, design and management in Australia

3.1 Introduction

Archaeological evidence has shown that Australian aboriginals have used artificial reefs for thousands of years, built from natural materials such as rock and wood (Carstairs 1988). Modern artificial reef construction however began in the 1960’s, sparked by work in the Virgin Islands and California, as well as early reviews of worldwide artificial reef developments (Pollard 1989). The trajectory of artificial reef development within Australia has been similar, albeit on a smaller scale to artificial reef development around the world, with early reefs consisting almost entirely of materials of opportunity, such as used tyres, vehicle bodies, scuttled vessels and concrete rubble (Pollard 1989, Kerr 1992).

Concerted scientific effort over the past 15 years, including a number of reef module trials, and input from global leaders in the field such as Japan and Korea, has provided valuable knowledge of the factors which influence the recruitment and succession of fish and epibiotic communities on artificial reefs, as well as the impact these structures have on ecological processes in the surrounding environment (Carr and Hixon 1997, Pitcher and Seaman Jr 2000, Department of Fisheries Western Australia 2010, Lowry et al. 2010). This knowledge, combined with more stringent legislative requirements, has triggered a surge in the use of modern purpose-built artificial reef modules within Australian waters, as they offer a number of significant benefits over reefs made from materials of opportunity (Pickering and Whitmarsh 1997, Department of Fisheries Western Australia 2010, Diplock 2010).

Previous reviews of artificial reefs in Australia e.g. Pollard and Matthews (1985), Kerr (1992), Branden et al. (1994), Coutin (2001), have provided information on early reef developments. As of 2001, the majority of artificial reefs within Australia were still made up of tyres (37%) or ships (22%) with only a small portion made from
concrete (6%) (Coutin 2001). However, limited work has been done on collating data on Australia’s most recent reef developments over the past 15 years, in which time a number of artificial reef programs have been developed, aimed at improving the quality and management of artificial reefs (Department of Fisheries Western Australia 2015, Fisheries Victoria 2015, New South Wales Department of Primary Industries 2015).

The aim of this chapter is to undertake a literature search to identify trends in artificial reef construction within Australia, since the deployment of the first artificial reef in 1965 to the present day. The chapter considers where and when artificial reefs were deployed, what the reefs were constructed from, and their primary purpose. It identifies trends in artificial reef design, construction, location and purpose, and assesses how these patterns have changed over the past 50 years.

### 3.2 Materials and Methods

This work builds on previous analyses of artificial reefs in Australia conducted by Pollard and Matthews (1985), Kerr (1992), Branden et al. (1994), Coutin (2001). It combines the data presented in those documents with those obtained during contemporary literature searches. These searches were conducted in search engines (e.g., Google and Google Scholar) and documents indexed in scientific databases (e.g., Scopus, Web of Science and Murdoch University). Keywords employed as search terms included “artificial reefs” and “habitat enhancement structures”, with additional words such as “Australia” and the names of Australia’s various coastal states and territories. For each reef, information such as the location, year of deployment, materials of construction (Table 3.1), purpose and builder/funder were obtained and stored in a database. Note that for the purpose of this report, the literature search was limited to purposely-placed benthic artificial reefs and thus accidental shipwrecks or floating Fish Aggregation Devices (FADs) have been excluded.
Table 3.1: Classification and description of materials used in the construction of artificial reefs.

<table>
<thead>
<tr>
<th>Materials of opportunity (MOP)</th>
<th>Description</th>
</tr>
</thead>
<tbody>
<tr>
<td>Tyres</td>
<td>Used vehicle tyres of any size.</td>
</tr>
<tr>
<td>Steel vessels</td>
<td>Steel hulled ships and other steel vessels that have been purposely scuttled for the creation of an artificial reef.</td>
</tr>
<tr>
<td>Rubble</td>
<td>Quarry rock and concrete rubble/waste purposely deployed to create an artificial reef.</td>
</tr>
<tr>
<td>Mixed MOP</td>
<td>Combination of two or more materials of opportunity at a single reef.</td>
</tr>
</tbody>
</table>

<table>
<thead>
<tr>
<th>Purpose-built</th>
<th>Description</th>
</tr>
</thead>
<tbody>
<tr>
<td>Concrete modules</td>
<td>Concrete modules of any size built specifically for use in the construction of an artificial reef <em>e.g.</em> concrete fish boxes and Reef Balls.</td>
</tr>
<tr>
<td>Steel modules</td>
<td>Steel modules of any size built specifically for use in the construction of an artificial reef, <em>e.g.</em> steel fish caves.</td>
</tr>
<tr>
<td>Geotextile bags</td>
<td>Geotextile bags, which can be filled with material such as sand, that have been specifically designed for use in artificial reef construction.</td>
</tr>
<tr>
<td>Mixed</td>
<td>Mixture of materials of opportunity and purpose-built modules at a single reef. Generally occurs when a reef is added to over multiple years.</td>
</tr>
</tbody>
</table>

3.3 Results and Discussion

To date, 121 artificial reefs were found to have been deployed in Australian waters (Fig. 3.1). While artificial reefs are present in each state and territory with a coastline, their numbers differ markedly. Relatively large numbers of artificial reefs were found in Victoria (28), South Australia (26), Queensland (22) and New South Wales (21), while lower numbers were present in Western Australia (11), Northern Territory (9) and Tasmania (4). The highest densities of artificial reefs were found close to major cities and/or within sheltered bays, such as the Gulf of St Vincent and Moreton Bay. In
contrast, remote coastlines, such as the Kimberley in Western Australia, and the Great Australian Bight, do not contain any artificial reefs with only a single reef present in the Gulf of Carpentaria (Fig. 3.1).

![Geographical distribution of artificial reefs (AR) in Australia. The numbers of artificial reefs in the waters of each state and territory are given in brackets.](image)

Currently, 65% of artificial reefs in Australia are composed from materials of opportunity. In some locations, such as South Australia and the Northern Territory, all artificial reefs deployed to date have been constructed from materials of opportunity (Fig. 3.2). Victoria is the only location to date where the proportion of purpose-built reefs is greater than those constructed from materials of opportunity.
Chapter 3

Fig. 3.2: The contribution of artificial reefs constructed from purpose-built material, materials of opportunity or both to the total number of artificial reefs in each state and territory and to Australia as a whole.

Among the purpose-built artificial reefs in Australia, the vast majority (34 out of 43) were found to be constructed from concrete modules, with only two reefs comprising steel modules and one of geotextile bags (Fig. 3.3). The constituency of reefs constructed from materials of opportunity was more diverse and included steel vessels (32), such as old warships, used tyres (28), as well as reefs constructed from a mix of waste materials (13). While rubble has been used, it has been so sparingly, with only five reefs purposely constructed to date from this material of opportunity (Fig. 3.3).
Australia’s first artificial reef was deployed in 1965 in Victoria (Fig 3.4). The reef was placed by the Victorian Department of Fisheries and Wildlife, who laid 400 tonnes of concrete pipes over an area of 4 hectares to create an artificial reef in Port Phillip Bay, with two additional steel vessels added in 1967. Although this reef initially provided good fishing for Pink Snapper (*Chrysophrys auratus*), it was placed on fine silt which slowly caused the pipes to sink (Kerr 1992). Since the deployment of this first reef, another 120 reefs have been constructed throughout Australia’s coastal states (Fig. 3.4).

The construction of reefs was found to fluctuate over the 50 year period with high numbers of reefs constructed between 1968-1973, 1982-1991 and 2009-2015 (Fig. 3.4). The highest number of reefs constructed in a single year was eight, which occurred in 1991 and 2011. While the construction of artificial reefs in some states, such as in Victoria, New South Wales and Western Australia, was spread out across the last 50 years, construction in South Australia and the Northern Territory occurred in distinct periods (Fig. 3.4). In the case of South Australia, deployment occurred almost exclusively between 1969 and 1973 and between 1983 and 1991, whereas construction in the Northern Territory occurred between 1982 and 1991, and 2011 and 2012.
Fig. 3.4: The number of artificial reefs deployed within Australia waters between 1965 and 2015, and the state or territory in which they were deployed.

The materials used to construct the various artificial reefs found in Australian waters differ among states and territories (Fig. 3.5). Tyres, for example, are the primary constituent of artificial reefs in South Australia, representing 18 out of 26 reefs, but have been used sparingly in other states. New South Wales, Victoria, Queensland and Western Australia have all invested significant effort in the deployment of purpose-built reefs, particularly concrete modules.

Fig. 3.5: The number of artificial reefs deployed in each of Australia’s coastal state and territories and the materials used to construct them.
Over the past 50 years there has been a clear shift in materials used to construct artificial reefs in Australia. Australia’s earliest artificial reefs were constructed predominantly from materials of opportunity, with the most abundant material being used tyres (Fig. 3.6). The use of materials of opportunity continued to be widely popular up until the early-90’s, however during the mid-80’s the relative proportion of reefs constructed from tyres decreased, and there was a switch to primarily the sinking of steel vessels. During the past 10 years however, artificial reefs have been constructed almost exclusively from purpose-built modules, primarily made from concrete (Fig. 3.6).

This early use of tyres for constructing artificial reefs within Australia likely reflects the availability and low cost of this material, and the view that this constituted recycling, whilst simultaneously creating additional habitat for fish and invertebrate communities (Sherman and Spieler 2006). Experience however has shown tyres to be unsuitable for use in artificial reef construction due a number of negative environmental impacts associated with their use in the marine environment, which have been described in Chapter 2.

Steel vessels, whilst being most popular during the mid-1980’s, have continued to be used in reef construction, with the latest reef of this type deployed in Australian waters in 2011 (ex-HMAS Adelaide). The continued use of these types of reef materials is due to their popularity with SCUBA divers, and provision of tourism opportunities. Whilst steel vessels continue to be used to construct artificial reefs, the methods of deploying these vessels within Australian waters has changed significantly. There are now more stringent clean up and safety requirements for scuttling steel vessels, which in Australia is regulated under the Environment Protection (Sea Dumping) Act 1981 (Department of the Environment 2015). Regulations designed to minimise negative environmental impact and the specific purpose of these reefs is likely to see them continue to be used in the future.

Although the first reef built from purpose-built concrete modules was employed in 1971, it was not until the 2000s, and particularly post 2010, that this type of material was widely used. Today it constitutes the dominant construction material for
contemporary artificial reefs and is more widely used than other purpose-built materials, such as steel modules and geotextile bags. Although there is an additional cost involved, these purpose-built reefs have shown to provide significant benefits over materials of opportunity that has been described in detail in Chapter 2.

The shift in artificial reef design over the past 10 years has been brought about by the growing awareness of artificial reef technology in Australia as well as the provision of additional funds towards reef programs from recreational fishing licences, as has been done in New South Wales, Victoria, and Western Australia (Department of Fisheries Western Australia 2015, Fisheries Victoria 2015, New South Wales Department of Primary Industries 2015).

The vast majority of artificial reefs currently deployed in Australian waters (96 out of 121) have been constructed for the primary purpose of enhancing fishing activities (Fig. 3.7). The next most common purpose for artificial reef deployment was for recreational SCUBA diving, with 19 such reefs present in Australia, many of which were organised by diving clubs themselves. Small numbers of reefs have also been deployed by scientists and industrial partners for research, as well as two artificial surfing reefs (Fig. 3.7). While a wide range of organisations, including community groups, fishing and diving clubs have deployed these structures, state and territory fisheries departments have installed ~65%.
Fig. 3.6: The number of artificial reefs deployed in Australian waters each year since 1965 and the materials used to construct them.
Fig. 3.7: Number of artificial reefs categorised by their primary purpose (noting that many have multiple purposes) and the groups that deployed the reefs over the past 50 years.

The focus on deploying artificial reefs for enhancing fishing activities is not surprising, as early reefs were often deployed and funded by commercial and recreational fishing groups, and more recently, funds from recreational fishing licences have been used by state fisheries to deploy artificial reefs (Kerr 1992, Department of Fisheries Western Australia 2015, New South Wales Department of Primary Industries 2015). Although these reefs may increase the presence of valuable recreational fish species, care is needed to ensure they do not increase the vulnerability of target species to over fishing.

Although much is known about the effect of fishing on fish stocks, there is still limited information available on the effect of artificial reefs on fish stocks (Mace 1997). Fish species likely to be attracted to artificial reefs will be similar to those on adjacent natural reefs, some of which may be slow growing and long-lived species, vulnerable to over fishing (Carr and Hixon 1997, Pickering and Whitmarsh 1997). To minimise negative impacts, the introduction of management plans which assess fish stocks and monitor the performance of artificial reefs is essential (Carr and Hixon 1997, Baine...
2001). Other recommendations for the management of artificial reefs include strict bag and size limits for fish, and an initial closure period for the reef to establish itself (Department of Fisheries Western Australia 2010). The effects on the surrounding environment such as tidal flow, wave action and sand movement should also be considered prior to the deployment of an artificial reef (Pickering and Whitmarsh 1997, Bortone et al. 2011).

### 3.4 Conclusion

Whilst Australia’s artificial reef developments have previously been behind those of other countries, the past 10 years has seen a surge in interest in the use of modern purpose-built artificial reefs (Pitcher and Seaman Jr 2000, Coutin 2001, Diplock 2010). These purpose-built reef modules offer significant benefits over materials of opportunity, and the availability of additional funds through recreational fishing licence fees has been successfully used in New South Wales, Victoria, and Western Australia to fund artificial reef programs and reduce pressure on natural reefs and could potentially be utilized by other states in the future.

As the vast majority of Australia’s artificial reefs have been deployed primarily for the purpose of enhancing recreational fishing, reefs have been deployed close to major cities and generally within popular fishing regions. Although this makes the reefs easily accessible, it also creates the potential for overfishing of target species. Future research should also aim to incorporate the socio-economic impacts of these structures and factors, such as reef visitation levels and catch rates, which have not been discussed in detail within this review. With the number of artificial reefs in Australia set to increase over the coming years, dedicated management and monitoring of these structures is essential (Carr and Hixon 1997, Pickering and Whitmarsh 1997).
Chapter 4: Observer bias in underwater video analysis

4.1 Introduction

Remote underwater video monitoring has been widely adopted for the non-destructive sampling of a broad range of organisms and environments (Somerton and Glendhill 2005, Harvey et al. 2013). It has been utilized in both shallow and deep-water marine environments and shown to be an effective method for comparing fish assemblages over large spatial scales (Stobart et al. 2007), assessing biodiversity (Malcolm et al. 2007, Harasti et al. 2015), monitoring marine protected areas (Cappo et al. 2003, Westera et al. 2003), and evaluating the effectiveness of artificial reefs (Folpp et al. 2011, Lowry et al. 2012).

Remote underwater video monitoring offers significant benefits over traditional diver visual census methods in that it reduces the need for skilled observers in the field and enables sampling of depths and for times not possible on SCUBA (Harding et al. 2000, Langlois et al. 2010, Lowry et al. 2012, Pelletier et al. 2012). The use of underwater video also has the additional benefit of providing a permanent data set, able to be retrieved at any time, allowing researchers access to a much wider suite of information (Cappo et al. 2003). Whilst this method enables the collection of large amounts of information in a relatively short time frame, it does have the limitation of requiring post-field video analysis to extract the data (Harvey et al. 2013). The processing, interpretation, image storage and retrieval of data can be a laborious task, which may result in a bottleneck of data analysis (Somerton and Glendhill 2005, Harvey et al. 2013).

As was explained in Chapter 1, Recfishwest is establishing a program for monitoring the fish faunas of two artificial reefs recently deployed off Bunbury and Dunsborough in Geographe Bay in south-western Australia. An essential part of this monitoring program is that it remains cost-effective. Recfishwest is therefore aiming to use recreational fishers, acting as citizen scientists, to deploy underwater cameras to collect footage that can be used to assess the characteristics of the fish faunas of these
reefs. However, one limitation of this strategy is that it requires people to extract data from the footage collected by the fishers.

As implied above, in order for Recfishwest's strategy to monitor the fish faunas of the Bunbury and Dunsborough reefs to be possible, it is essential to find a cost-effective means for extracting data from the underwater video footage collected by the fishers. One possible solution that has been suggested is to get university students to extract information from video footage as part of their studies.

Whilst this method may counter the problems associated with data extraction, there is the potential for observer bias, as a number of different students will be involved in extracting data from the footage. Observer bias has the potential to render the data on fish faunas of the artificial reefs obtained via the footage collected by recreational fishers useless, as it could confound differences between observers with real spatial and temporal effects (Thompson and Mapstone 1998). It is therefore important to provide some assessment of the potential for observer bias in extracting data on fish faunas from such footage. The overall goal of this study was to make such an assessment.

The first specific aim of this study was to determine what level of observer bias, if any, is present among the observers when extracting the following information about fishes captured on remotely collected underwater footage; (i) the relative abundance (MaxN), (ii) species richness and (iii) species composition. Since observer bias was detected, the second aim was to develop a series of recommendations that can be implemented to reduce observer effects in the context of using university students to extract data from underwater video footage collected by recreational fishers.
4.2 Materials and Methods

4.2.1 Study Site

4.2.1.1 Geographe Bay

Geographe Bay is the southern most protected marine embayment in south-west Australia, with a low energy, but dynamic sandy coastline. The bay covers an area of roughly 290 square nautical miles, ranging from the north-west point of Cape Naturaliste (33° 32’ S, 115°00’E), to the Bunbury breakwater (33° 18’S, 115° 39’ E) (Bellchambers et al. 2006). This position gives the bay a northerly aspect with a predominately west to east longshore drift. The bay has a maximum depth of 30 m and normally experiences a semidiurnal tidal range of ~0.5 m (Bellchambers et al. 2006).

The substratum in Geographe Bay slopes gently seaward (~2 m km$^{-1}$) and is dominated by expansive areas (~70%) of monospecific seagrass meadows, comprised predominantly of Posidonia sinuosa, and peripheral assemblages of Amphibolis Antarctica (Walker et al. 1987, Mcmahon et al. 1997). The area experiences a Mediterranean climate, characterized by warm, dry summers and cool, wet winters. The average annual rainfall of the area is approximately 806 mm, with the majority (85%) falling between May and October (Walter 1973).

4.2.1.2 Geographe Bay artificial reefs

Each of the Geographe Bay artificial reefs consist of 30 ten-tonne reinforced concrete ‘Fish Box’ modules, placed in clusters of five, which together cover area of approximately four hectares (Fig 4.1: Department of Fisheries Western Australia 2015). The reef modules were designed by HaeJoo, and contain curved cross braces to promote the upwelling of nutrients (HaeJoo 2015b). These reefs were deployed specifically to enhance recreational fishing within the bay and, in particular, to increase the abundance of target species such as Pink Snapper (Chrysophrys auratus), Samson Fish (Seriola hippos) and Silver Trevally (Pseudocaranx dentex). Both reefs are located close to towns, i.e. Dunsborough and Bunbury, and were deployed within 5 km of the nearest
boat ramp. The Dunsborough reef is located at 115° 9.980’ E, 33° 33.962’ S, in 27 m of water, and the Bunbury reef is located at 115° 35.900’ E, 33° 18.500’ S at a depth of 17 m (Department of Fisheries Western Australia 2015).

![Fig. 4.1. The artificial reef deployment sites and cluster orientations. (Bottom right insert) The artificial reef ‘Fish Box’ module designed by HaeJoo. Reproduced from Department of Fisheries Western Australia (2015).](image)

### 4.2.2. Source of data

All underwater video footage employed in this study was collected from the Dunsborough artificial reef during two sampling trips on the 10th and 19th of March 2015 using a Baited Remote Underwater Video (BRUV) system. The data collection, design of the sampling regime and the construction of the BRUV system, was performed solely by Ecotone consulting and Recfishwest with no input from staff or students from Murdoch University.
4.2.3. Sampling regime

The sampling method performed by Ecotone consulting and Recfishwest involved the haphazard dropping of BRUVs in the vicinity of the artificial reef modules using GPS for navigation. Each drop involved positioning the boat above a reef module and lowering the BRUV over the boat until it reached the sea floor. Camera submersion times averaged ~20 minutes. Upon retrieval, the video footage was extracted from the camera and the BRUV reset and rebaited before being deployed at a new location (Florisson 2015).

4.2.4 BRUV design

The frame of the BRUV was constructed from class 9 Polyvinyl Chloride (PVC) irrigation pipe, which is rated to 8.88 atmospheres and able to withstand pressures associated with water depths of up to ~80 m (Fig. 4.2). Sections of PVC pipe and pipe connections were glued together with PVC cement suitable for use on pressurized water pipes. The frame of the BRUV was stabilised by two skids, each filled with four, 680g lead weights, making the BRUV negatively buoyant, and giving it a total weight of 5.5 kg.

The bait arm of the BRUV was designed to be suspended 150 mm above the substrate with a length of 600 mm from the central point of the BRUV. The bait bag, which measured 180 mm x 100 mm, was constructed from plastic mesh and positioned
500 mm in front of the camera. Before each deployment, 500g of Sardine (*Sardinops* spp.) was placed inside the bait bag and securely fastened. This bait is widely used in BRUV studies as its soft, oily flesh has shown to be an effective attractant for fish (Stobart et al. 2007, Harvey et al. 2013, Mallet and Pelletier 2014).

Finally a GoPro Hero 4 Silver Action Video Camera™, placed in a waterproof housing with an additional Battery BacPac™, was mounted to the BRUV. This camera was chosen due to its small size and simple design, making it easy to work with whilst still providing high definition video footage, *i.e.* 1080p resolution at a frame rate of 60 frames per second (GoPro 2015).

### 4.2.5 Observers and video analysis

A total of four observers took part in this study. Each observer was required to be a recreational fisher who engaged in fishing activities at least once a month, and had completed a Bachelor of Science majoring in Marine Science in the past three years from Murdoch University. The four observers in his study included two volunteers, one university student who had logged data from the Recfishwest video footage as part of their university studies and the author. Whilst this study would have benefited from additional observers, limited funding and time constraints due to the availability of the video footage and the time it took each volunteer to watch the required amount of footage only allowed data from four observers to be obtained and analysed.

Prior to analysis, the provided raw videos were coded according to the trip collection date (t), camera number (c), and video data number. For example, a video collected on trip one, by camera one, with a video data number of 0001, would be coded (t1c1-0001). Two additional factors were given to each video that indicated the camera direction as facing reefs modules (F) or not facing reef modules (NF), as well as a unique observer number between 1 and 4.

Previous work by Florisson (2015) identified significant differences in the composition of fish species depending on whether the footage was collected facing or
not facing reef modules. Thus, whilst not being the main focus of this study, this factor was considered and incorporated into the statistical modelling.

Each observer was provided with the same set of 30 separate videos collected from the Dunsborough artificial reef by Recfishwest and Ecotone Consulting using BRUVs. Observers were instructed to analyse each video for a total of 5 minutes, between the allocated time slot of 7-12 minutes, giving a total of 150 minutes of footage analysed by each observer. Observers were given no species identification training but were provided with a copy of “Sea Fishes of Southern Australia” by Hutchins and Swainston (1986), as well as a number of links to online taxonomic data bases to assist in species identification.

Analysis of each video involved identifying each fish to the lowest possible taxonomic level and providing an index of its relative abundance, namely MaxN. MaxN is defined as the maximum number of individuals of each species observed in a single frame in the footage being analysed. MaxN is a widely used index in underwater video studies and provides a conservative measure of relative abundance that eliminates the chance of double counting (Willis and Babcock 2000, Cappo et al. 2003, Watson 2006). Whilst is not classified as a fish, *Sepioteuthis australis* (Southern Calamari), has been included within this study as it is an important recreational species with the Geographe Bay area and heavily targeted by fishers.

All video footage was reviewed using the multimedia program QuickTime. Abundance data from each observer were compiled into a single data matrix where each video had a unique identifier code as well as additional factors that indicated the observer and the camera direction. All following statistical analysis was performed from this single data matrix.
4.2.6 Statistical analyses

All statistical analyses were undertaken using the Primer v7 multivariate statistics software package, with the PERMANOVA+ add on (Anderson et al. 2008, Clarke and Gorley 2015). For all analyses, the null hypothesis of no significant difference between a priori groups was rejected if the significance level (p) was ≤ 0.05.

4.2.6.1 Univariate analyses

Two-way Permutational Multivariate Analysis of Variance (PERMANOVA; Anderson et al., 2008) was employed to determine whether the values for taxon richness (i.e. the number of taxa) and total MaxN (i.e. the sum of the MaxN values for each species in a sample) differed between observers and camera positions (facing towards and away from the artificial reef). Both of these variables were considered fixed. The DIVERSE routine was used to calculate, for each individual sample, the taxon richness and total MaxN.

Prior to subjecting the data for each dependent variable to two-way PERMANOVA, the extent of the linear relationship between the loge-transformed mean and loge-transformed standard deviation for each of the various sets of replicate samples for both variables was examined. This approach was used to determine whether the data for each variable required transformation to meet the test assumption of homogenous dispersions among a priori groups and, if so, to identify the appropriate transformation required (Clarke et al. 2014a). This analysis demonstrated that taxon richness required no transformation, whilst total MaxN required a fourth root transformation.

The pre-treated data, where required for each variable, were then used to construct separate Euclidean distance matrices and subjected to two-way PERMANOVA. Graphs of the transformed arithmetic means and associated ± 95% confidence intervals were plotted to visualise the extent of any differences between the main effects and/or interactions, noting that trends between observers are the main focus of this study.
4.2.6.2 Multivariate analysis

PERMANOVA, Analysis of Similarities (ANOSIM; Clarke and Green 1988) non-metric Multi-Dimensional Scaling (nMDS) ordination plots (Clarke 1993) and shade plots (Clarke et al. 2014b, Tweedley et al. 2015) were employed to elucidate whether the composition of the fish and cephalopod faunas identified on the BRUV footage differed between observers and camera positions and, if so, the species that were responsible for those differences.

The MaxN for each species in each individual sample was subjected to a fourth root transformation to down weigh the contributions of highly abundant taxa and balance them with those of less abundant taxa. These transformed data were then used to construct a Bay-Curtis similarity matrix and subjected to the same two-way PERMANOVA test described above for taxa richness and total MaxN, only this time employing multivariate data. However, in this instance, the sole purpose of the PERMANOVA was to determine if there was an interaction between the site and camera position main effects and, if so, to determine the extent of those interactions relative to each other and to those of the main effects (Lek et al. 2011). If the interaction was not significant, or relatively small in relation to the main effects, the matrix was then subjected to a two-way ANOSIM test. ANOSIM was preferred at this stage of the analysis because, unlike PERMANOVA, this test is fully non-parametric and thus more robust, and because the ANOSIM R-statistic provides a universal measure of group separation to test for significant interactions between region and position (Lek et al. 2011). The magnitude of the $R$ statistic typically ranges between 1, when the compositions of the samples within each group are more similar to each other than to that of any of the samples from other groups, down to ~0, when within-group and between-group similarities do not differ (Clarke and Gorley 2015).

The same Bray-Curtis similarity matrix was then subjected to nMDS to produce an ordination plot, which provided a visual representation of the trends in faunal composition among observers. However, as this plot showed the position of all 120 samples it was hard to interpret accurately the trends among a priori groups. Therefore, a second nMDS plot was constructed, only this time from a distance among the
centroids matrix. This matrix creates averages in the ‘Bray–Curtis space’ calculated from the groups of replicate samples, in this case averages of each observers videos from a single camera direction thus condensing the 120 samples into eight (Anderson et al. 2008). These plots, which show low-dimensional approximations to the pattern of group centroids in the full-dimensional space, are subsequently referred to as centroid nMDS ordination plots (Lek et al. 2011).

Finally, shade plots were employed to produce a visual display of the abundance matrix of variables (transformed and standardized species counts) against samples (groups of videos). As the PERMANOVA test demonstrated that the species composition differed among both observers and camera position, but that the interaction between these factors was not significant, the fourth-root transformed MaxN data for each species in each sample was averaged and used to create two data matrices. In the first the transformed data was averaged across the four observers and in the second it was averaged across the two camera positions. The data in these two matrices were standardized and subjected to the Shade plot routine. This produced a visual display of the abundance matrix of variables (transformed and standardized species counts) against samples (either observers or camera positions), where the white represents the absence of taxa in a sample and the intensity of grey-scale shading is linearly proportional to ‘abundance’ (Clarke et al. 2014b). The taxa (y axis of the shade plot) are ordered to optimise the seriation statistic (ρ) by non-parametrically correlating their resemblances to the distance structure of a linear sequence (Clarke et al. 2014a). This seriation was constrained by the family of the taxa so that taxa within the same family, regardless of their similarity to one another, were kept together and separate from other families. The order of both the samples (displayed on the x axis) in the case of the shade plot showing observers were determined independently by the results of a group-average hierarchical agglomerative cluster analyses employing resemblance matrices defined using Whittaker’s index of association (Whittaker 1952, Valesini et al. 2014).
4.3. Results

The four observers identified a combined total of 46 taxa to species, three to genus and three to family (Table 4.1). The greatest number of taxa identified by a single observer was 36 (Observer 4), while the lowest number of taxa identified was 26 (Observer 3). Observer 4 recorded the highest total mean MaxN count, i.e. 34.1, while the mean MaxN counts for the other three observers ranged from 27 and 30 (Table 4.1).

All observers identified Pseudocaranx spp. and Coris auricularis as the first and second most abundant taxa. These two taxa dominated the data set and were found to make up ~70 % of the individuals identified by all observers. Neatypus obliquus was identified as the third most abundant species by Observers 1, 2 and 3, whilst the third most abundant species identified by Observer 4 was Trachurus novaezelandiae.

Thirteen of the species detected by Observers 1, 2 and 4 were not identified by Observer 3, including species such as T. novaezelandiae, Parequula melbournensis, and Austrolabrus maculatus. However, Observer 3 identified eight species that were not detected by any other observer, including Caesioscorpis theagenes and Labroides dimidiatu. Meuschenia freycineti was only identified by Observer 1, and Observer 2 was the only observer to identify Eubalichthys mosaicus, Cheilodactylus nigripe, Halichoeres brownfieldi and Lagocephalus lunaris.
<table>
<thead>
<tr>
<th>Species</th>
<th>Total</th>
<th>Observer 1</th>
<th>Observer 2</th>
<th>Observer 3</th>
<th>Observer 4</th>
</tr>
</thead>
<tbody>
<tr>
<td>Pseudocaranx spp.</td>
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<td>3.97</td>
</tr>
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<td>0.07</td>
<td>1.4</td>
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<td>0.69</td>
<td>0.9</td>
<td>9</td>
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</tr>
<tr>
<td>Myliobatis australis</td>
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<td>0.04</td>
<td>0.6</td>
<td>7</td>
<td>0.20</td>
</tr>
<tr>
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<td>0.6</td>
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<tr>
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<td>0.04</td>
<td>0.5</td>
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<td>0.4</td>
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<td>0.4</td>
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<td>0.04</td>
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<tr>
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<td>Caepioscorps theagenes</td>
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<td>0.08</td>
<td>0.3</td>
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<td>0.17</td>
</tr>
<tr>
<td>Labroides dimidiatae</td>
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<td>0.01</td>
<td>0.3</td>
<td>21</td>
<td>0.13</td>
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<tr>
<td>Lagocephalus scelerata</td>
<td>0.10</td>
<td>0.03</td>
<td>0.3</td>
<td>21</td>
<td>0.13</td>
</tr>
<tr>
<td>Cheilodactylus vestitus</td>
<td>0.09</td>
<td>0.01</td>
<td>0.3</td>
<td>23</td>
<td>0.07</td>
</tr>
<tr>
<td>Chelmonops curvisus</td>
<td>0.08</td>
<td>0.03</td>
<td>0.2</td>
<td>24</td>
<td>0.10</td>
</tr>
<tr>
<td>Nemadactylus macropterus</td>
<td>0.08</td>
<td>0.05</td>
<td>0.2</td>
<td>24</td>
<td>0.10</td>
</tr>
<tr>
<td>Ophthalomelops lineolatus</td>
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<td>0.03</td>
<td>0.2</td>
<td>26</td>
<td>0.03</td>
</tr>
<tr>
<td>Pentapodus vitta</td>
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<td>0.06</td>
<td>0.2</td>
<td>27</td>
<td>0.03</td>
</tr>
<tr>
<td>Papercorpus recurvisrostris</td>
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<td>0.02</td>
<td>0.2</td>
<td>27</td>
<td>0.07</td>
</tr>
<tr>
<td>Chrysemys arakens</td>
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<td>0.02</td>
<td>0.1</td>
<td>29</td>
<td>0.07</td>
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<td>Pictilabrus latilatus</td>
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<td>0.04</td>
<td>0.1</td>
<td>30</td>
<td>0.17</td>
</tr>
<tr>
<td>Anoplograpus lenticulareis</td>
<td>0.04</td>
<td>0.04</td>
<td>0.1</td>
<td>30</td>
<td>0.10</td>
</tr>
<tr>
<td>Parapercis ramsayi</td>
<td>0.04</td>
<td>0.03</td>
<td>0.1</td>
<td>30</td>
<td>0.03</td>
</tr>
<tr>
<td>Nemadactylus veniliaesi</td>
<td>0.03</td>
<td>0.03</td>
<td>0.1</td>
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<td>Parauzonochitius hutchinsi</td>
<td>0.03</td>
<td>0.02</td>
<td>0.1</td>
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<td>0.03</td>
</tr>
<tr>
<td>Lagocephalus sp.</td>
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<td>0.01</td>
<td>0.1</td>
<td>33</td>
<td>0.13</td>
</tr>
<tr>
<td>Suezichthys cyanolaemus</td>
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<td>0.29</td>
<td>0.8</td>
<td>36</td>
<td>0.03</td>
</tr>
</tbody>
</table>
Table 4.1 continued: Species table showing the mean MaxN (X) and standard error (SE) of each of the 52 fish and cephalopod taxa recorded by each of four observers who analysed the same five minute portion of the same 30 videos recorded using BRUV on the Dunsborough artificial reef. For each taxon, a percentage contribution (%) and ranking by mean MaxN (R) was calculated. Abundant species *i.e.* those that contributed ≥ 5 % to abundance recorded by any observer are shaded in grey.

<table>
<thead>
<tr>
<th>Species</th>
<th>X</th>
<th>SE</th>
<th>%</th>
<th>R</th>
<th>X</th>
<th>SE</th>
<th>%</th>
<th>R</th>
<th>X</th>
<th>SE</th>
<th>%</th>
<th>R</th>
<th>X</th>
<th>SE</th>
<th>%</th>
<th>R</th>
<th>X</th>
<th>SE</th>
<th>%</th>
<th>R</th>
</tr>
</thead>
<tbody>
<tr>
<td>Pseudolabrus biserialis</td>
<td>0.03</td>
<td>0.062</td>
<td>0.08</td>
<td>36</td>
<td>0.03</td>
<td>0.03</td>
<td>0.12</td>
<td>26</td>
<td>0.03</td>
<td>0.03</td>
<td>0.11</td>
<td>28</td>
<td>0.03</td>
<td>0.03</td>
<td>0.10</td>
<td>27</td>
<td>0.03</td>
<td>0.03</td>
<td>0.10</td>
<td>27</td>
</tr>
<tr>
<td>LABRIDAE spp.</td>
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<td>0.018</td>
<td>0.08</td>
<td>36</td>
<td>0.03</td>
<td>0.03</td>
<td>0.12</td>
<td>26</td>
<td>0.03</td>
<td>0.03</td>
<td>0.11</td>
<td>28</td>
<td>0.03</td>
<td>0.03</td>
<td>0.10</td>
<td>27</td>
<td>0.03</td>
<td>0.03</td>
<td>0.10</td>
<td>27</td>
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<td>Neosebastes pandus</td>
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<td>0.014</td>
<td>0.08</td>
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<td>0.03</td>
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<td>26</td>
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<td>28</td>
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<td>0.03</td>
<td>0.10</td>
<td>27</td>
<td>0.03</td>
<td>0.03</td>
<td>0.10</td>
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<td>Lagocephalus lunaris</td>
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<td>0.08</td>
<td>36</td>
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<td>0.03</td>
<td>0.12</td>
<td>26</td>
<td>0.03</td>
<td>0.03</td>
<td>0.11</td>
<td>28</td>
<td>0.03</td>
<td>0.03</td>
<td>0.10</td>
<td>27</td>
<td>0.03</td>
<td>0.03</td>
<td>0.10</td>
<td>27</td>
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<tr>
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<td>23</td>
<td>0.07</td>
<td>0.05</td>
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<td>Halichoeres brownfieldi</td>
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<td>0.017</td>
<td>0.06</td>
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<td>0.07</td>
<td>0.05</td>
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<td>0.23</td>
<td>23</td>
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<td>Achoerodus gouldii</td>
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<td>0.23</td>
<td>23</td>
<td>0.07</td>
<td>0.05</td>
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<td>0.23</td>
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<td>0.012</td>
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<td>0.07</td>
<td>0.05</td>
<td>0.23</td>
<td>23</td>
<td>0.07</td>
<td>0.05</td>
<td>0.23</td>
<td>23</td>
<td>0.07</td>
<td>0.05</td>
<td>0.23</td>
<td>23</td>
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</tr>
<tr>
<td>Platyrocephalus spp.</td>
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<td>0.012</td>
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<td>0.03</td>
<td>0.03</td>
<td>0.12</td>
<td>26</td>
<td>0.03</td>
<td>0.03</td>
<td>0.12</td>
<td>26</td>
<td>0.07</td>
<td>0.05</td>
<td>0.24</td>
<td>17</td>
<td>0.07</td>
<td>0.05</td>
<td>0.24</td>
<td>17</td>
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<tr>
<td>Scorpaenodes smithi</td>
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<td>0.012</td>
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<td>0.03</td>
<td>0.12</td>
<td>26</td>
<td>0.03</td>
<td>0.03</td>
<td>0.12</td>
<td>26</td>
<td>0.07</td>
<td>0.05</td>
<td>0.24</td>
<td>17</td>
<td>0.07</td>
<td>0.05</td>
<td>0.24</td>
<td>17</td>
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<tr>
<td>Trygonoptera ovalis</td>
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<td>0.03</td>
<td>0.12</td>
<td>26</td>
<td>0.03</td>
<td>0.03</td>
<td>0.12</td>
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<td>0.03</td>
<td>0.12</td>
<td>26</td>
<td>0.03</td>
<td>0.03</td>
<td>0.12</td>
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<td>0.03</td>
<td>0.12</td>
<td>26</td>
<td>0.03</td>
<td>0.03</td>
<td>0.12</td>
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<td>0.03</td>
<td>0.03</td>
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<tr>
<td>MONACANTHIDAE spp.</td>
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<td>0.008</td>
<td>0.03</td>
<td>49</td>
<td>0.03</td>
<td>0.03</td>
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<td>0.03</td>
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<td>0.03</td>
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<td>0.03</td>
<td>0.03</td>
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</tr>
<tr>
<td>OSTRACIIDAE spp.</td>
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<td>49</td>
<td>0.03</td>
<td>0.03</td>
<td>0.12</td>
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</tr>
<tr>
<td>Trygonoptera personata</td>
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<td>0.03</td>
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<td>0.03</td>
<td>0.11</td>
<td>28</td>
<td>0.03</td>
<td>0.03</td>
<td>0.11</td>
<td>28</td>
<td>0.03</td>
<td>0.03</td>
<td>0.11</td>
<td>28</td>
<td>0.03</td>
<td>0.03</td>
<td>0.11</td>
<td>28</td>
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<tr>
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<td>34</td>
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<td></td>
<td></td>
<td>36</td>
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</tr>
<tr>
<td>Total mean MaxN</td>
<td>29.9</td>
<td>28.70</td>
<td>29.50</td>
<td>27.33</td>
<td>34.10</td>
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</tr>
</tbody>
</table>

Number: 52, 32, 34, 26, 36
Total mean MaxN: 29.9, 28.70, 29.50, 27.33, 34.10
### 4.3.1 Univariate analysis

Whilst there was slight variation, PERMANOVA showed no significant difference between either the mean number of species (Table 4.2A; Fig. 4.3A), or the relative abundance of species (Table 4.2B; Fig. 4.3B) identified per sample between observers. Significant differences were detected between the number of species on reef facing and not reef facing camera footage (Table 4.2A). Observers 2 and 4 identified the most species per sample, averaging just over 6 species, whilst the lowest mean number of species identified per sample was 5 (Observer 3). The highest mean abundance was recorded by Observer 4, with a mean of ~30, with the lowest recorded by Observer 3 with a mean of ~26.

**Table 4.2:** Mean squares (MS), Pseudo-\(F\) (\(pF\)) values and significance levels (\(P\)) for a two-way PERMANOVA test on (A) number of species, between observers and camera position and (B) abundance (total MaxN) counts between observers and camera position.

<table>
<thead>
<tr>
<th>(A) Number of species</th>
<th>df</th>
<th>MS</th>
<th>(pF)</th>
<th>(P)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Observer</td>
<td>3</td>
<td>11</td>
<td>1.88</td>
<td>0.148</td>
</tr>
<tr>
<td>Position</td>
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<td>64.53</td>
<td>11</td>
<td>0.003</td>
</tr>
<tr>
<td>Residual</td>
<td>112</td>
<td>656.9</td>
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<td></td>
</tr>
</tbody>
</table>

<table>
<thead>
<tr>
<th>(B) Abundance</th>
<th>df</th>
<th>MS</th>
<th>(pF)</th>
<th>(P)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Observer</td>
<td>3</td>
<td>0.045</td>
<td>0.68</td>
<td>0.55</td>
</tr>
<tr>
<td>Position</td>
<td>1</td>
<td>0.089</td>
<td>1.35</td>
<td>0.23</td>
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<tr>
<td>Residual</td>
<td>112</td>
<td>7.41</td>
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</table>
Fig 4.3. (A) Mean number of species identified per sample by each observer and (B) the fourth root transformed, total mean MaxN identified per sample by each observer. Error bars represent 95% confidence intervals.
**4.3.2 Multivariate analysis**

PERMANOVA demonstrated that the composition of species identified by the four observers differed significantly (Table 4.3). ANOSIM found that the data collected by Observers 1, 2 and 4 were not significantly different, but were invariably significantly different to the data collected by Observer 3 (Table 4.3). These trends are highlighted in the 3-dimentional nMDS plot that shows a clear grouping of samples from Observer 3, whilst the remaining three observer samples show no clear pattern (Fig. 4.5A). Significant differences in species composition were also detected by observers between footage from reef facing and not reef facing samples (Table 4.3). This is shown visually in the nMDS centroid plot that shows clear grouping of facing and not facing samples by all observers as well as a close grouping between Observers 1,2 and 4 (Fig. 4.5B).

**Table 4.3**: Mean squares (MS), Pseudo-F (pF) values and significance levels (P) for a two-way PERMANOVA test on the species composition between observers and camera position.

<table>
<thead>
<tr>
<th>Species composition</th>
<th>df</th>
<th>MS</th>
<th>pF</th>
<th>P</th>
</tr>
</thead>
<tbody>
<tr>
<td>Observer</td>
<td>3</td>
<td>3145</td>
<td>2.46</td>
<td>0.002</td>
</tr>
<tr>
<td>Position</td>
<td>1</td>
<td>11115</td>
<td>8.69</td>
<td>0.001</td>
</tr>
<tr>
<td>Observer x Position</td>
<td>3</td>
<td>0.83</td>
<td>0.66</td>
<td>0.83</td>
</tr>
<tr>
<td>Residual</td>
<td>112</td>
<td>1.43E+05</td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

**Table 4.4**: The coefficient of correlation (R) and significance levels (P) for ANOSIM analysis results of the species composition between observers. Significant differences indicated in bold.

<table>
<thead>
<tr>
<th>Observer</th>
<th>R</th>
<th>P</th>
</tr>
</thead>
<tbody>
<tr>
<td>1 vs 2</td>
<td>-0.049</td>
<td>0.976</td>
</tr>
<tr>
<td>1 vs 3</td>
<td><strong>0.141</strong></td>
<td><strong>0.001</strong></td>
</tr>
<tr>
<td>1 vs 4</td>
<td>-0.055</td>
<td>0.986</td>
</tr>
<tr>
<td>2 vs 3</td>
<td><strong>0.174</strong></td>
<td><strong>0.001</strong></td>
</tr>
<tr>
<td>2 vs 4</td>
<td>-0.042</td>
<td>0.949</td>
</tr>
<tr>
<td>3 vs 4</td>
<td><strong>0.185</strong></td>
<td><strong>0.001</strong></td>
</tr>
</tbody>
</table>
Fig 4.5. A.) A 3d nMDS plot constructed using the Bay-Curtis Similarity matrix, using fourth root transformed data of the MaxN for each species in each sample coded by observer. B.) A 2d centroid nMDS ordination plot, derived from distance among centroid matrices constructed from the Bay-Curtis Similarity matrix, created using fourth root transformed data of the MaxN for each species in each sample coded for observer.

A shade plot showing the mean MaxN of species identified highlights trends in species and families identified between the four observers (Fig. 4.6). *Pseudocaranx* spp., *Anoplocapros amygdaloides* and *Coris auricularis*, dominated the data set and were found in similarly high abundance by all observers. Other species found in similar
abundance by all four observers were *Neatypus obliquus*, *Myliobatis australis*, and *Glaucosoma hebraicum*. A hierachical conglomerative cluster analysis of the similarity between observers showed that the species composition of Observers 1 and 4 had the highest similarity (91%). This was followed by Observer 2, who showed a similarity of 89% to Observers 1 and 4, whilst Observer 3 showed the lowest similarity to the other observers with a species composition similarity of 70% (Fig 4.6). Variation between Observer 3 and the other observers was found to be highest for taxa within the families Labridae, Cheilodactylidae and Monacanthidae (Fig 4.6).

Fig 4.6: Shade plot illustrating the fourth root transformed relative abundance (MaxN) of species with shading intensity being proportional to abundance. Relative abundance (MaxN) counts are categorized by observer, and species are ordered by their family.
As with the shade plot comparing the species composition between observers, a shade plot showing the species composition between reef facing and not reef facing footage highlights that a small number of species dominated the data set and comprised the majority of individuals (Fig 4.7). Overall the relative abundance and number of species was found to be higher on footage that was collected facing the reef modules. Whilst the most abundant species *Pseudocaranx* spp., was found to be in similar densities on both facing and not facing footage, *Coris auricularis* and *Anoplacopros amygdaloides* were found in higher densities on facing footage (Fig. 4.7).

![Shade plot illustrating the fourth root transformed relative abundance (MaxN) of species with shading intensity being proportional to abundance. Relative abundance (MaxN) counts are categorized by facing (F) and not facing (NF) camera positions, and species are ordered by their family.](image_url)
4.4 Discussion

The detection and management of observer bias is key to maintaining the quality of data collected in any monitoring study (Harding et al. 2000, Pattengill-Semmens and Semmens 2003, Williams et al. 2006). This study has provided a preliminary assessment of the extent of bias among four observers in extracting data on the abundance and composition of fish from underwater footage of an artificial reef deployed off Dunsborough. The study found that, whilst the fish fauna data extracted from the footage by three of the observers were similar, there was significant variation between the results obtained by these three observers and those obtained by a fourth observer (Observer 3). Whilst the difference between the numbers of species or the number of individuals identified among the four observers were not statistically significant, there was a significant difference in the overall species composition. This indicates that individual fish on the footage were misidentified in some cases, particularly by Observer 3, rather than unsighted.

Abundance estimates of C. auricularis were fairly consistent across all observers, however, there was strong variation in the abundance of other Labridae species. Past studies have shown that species within the family Labridae are particularly difficult to identify, and labrids have been a primary source of error with less experienced observers (Williams et al. 2006). This is likely due not only to the physical similarity of many of these species but also their tendency to hide among structures and vegetation (Hutchins and Swainston 1986, Froese and Pauly 2015).

Differences were also seen within the family Carangidae, particularly in the abundance of T. novaezelandiae. Species within the family Carangidae have also been previously difficult to identify due to the fast moving, schooling behaviour of some of these species (Thresher and Gunn 1986). It is possible that variation in the abundance of T. novaezelandiae was due to confusion with Pseudocaranx spp., which was identified in high numbers by all observers. These two taxa show similar behavioural characteristics and colour markings, and could be easily confused if both are present in a fast moving school (Hutchins and Swainston 1986). Species within the family Monacanthidae also showed variation across observers. These species also exhibit...
similar behaviors and colour between species and are potentially confused by observers who are not familiar with the species (Hutchins and Swainston 1986).

Although this study has focused primarily on the detection of observer bias, it has also been noted that similar to previous work by Florisson (2015), all observers identified significant differences between the species composition on facing and not facing footage. This is likely due to habitat preference between different species, as well as the increased availability of food and shelter provided by the artificial reefs. Previous studies have shown that species abundance was greater on artificial reefs than the surrounding area and it is possible that the additional shelter and habitat created by the Geographe Bay artificial reefs promotes an increased abundance of fish species (Sherman et al. 2002, Folpp et al. 2011). However the limited data available means only assumptions can be made, and further investigation is required to determine the effects that camera apposition has on assessing the fish fauna of artificial reefs and if this should be taken into consideration in future monitoring.

Reducing observer bias in future studies

The limited taxonomic experience of observes and familiarity with species that were present on the video footage is likely a key cause of the variation between observers. Although all observers had similar educational qualifications and were recreational fishers, observer bias was still present. The provision of additional experience through observer training has shown to be an effective method of reducing bias (Thompson and Mapstone 1998). Previous studies of observer bias in underwater visual census by divers have shown that with experience, observer bias rapidly diminishes and only minor variation is present between well trained individuals (Williams et al. 2006, Yoklavich and O'Connell 2008).

Training of individuals to conduct video analysis should be done using a range of environments and organisms likely to be encountered, using footage that has been previously reviewed by an experienced observer (Tissot 2008). Initially, inexperienced observers should be guided through a number of videos and issues of identification
should be discussed as they arise. Once observers begin to log information on their own, these data can be quantitatively compared to those of a more experienced observer to detect the level of variation. Tissot (2008) recommends a minimum similarity of 90% between observers before individuals can be left to conduct their own analysis.

Providing observers with the opportunity to have species identifications reviewed by a more experienced observer/taxonomist would help to increase the quality of data. One of the key benefits of using underwater video is the ability to view the footage multiple times if ever there is confusion with the identification of a species. This can be easily achieved by having observers take snapshots from the footage of a species they were unclear on the identification of and send it to a reviewer. These images could then be used to create a database over time that could be used as a reference in future monitoring of reefs in southwest Western Australia.

Another method of potentially reducing observer bias is by focusing the analysis on a narrower range of taxa (Thresher and Gunn 1986, Williams et al. 2006). As this study included all species present in the field of view, observers may have been overwhelmed at times with large numbers of fish and species occurring simultaneously, and miss cryptic or less common species (Smith 1989, Samoilys and Carlos 2000). As the south-west artificial reefs were deployed primarily to increase the abundance of target recreational fishing species, analysis of footage could focus primarily on the abundance of recreational species such as Chrysophrys auratus and Seriola hippos, to provide better abundance estimates on these key species, as well as reduce the time taken to analyse footage.

Varying water clarity and light can also affect the ability to identify species and provide accurate measurements of relative abundance (MaxN). Harasti et al. (2015) found that standardizing the field of view to approximately 2 m behind the bait bag significantly reduced the effects of water visibility. This can be estimated visually by the observer, by ensuring the bait bag is a set length e.g. 1 m, and using it as a reference.
Conclusion

Species identification appeared to be the primary source of variation among observers, particularly of species within the family Labridae, Carangidae and Monacanthidae. Fortunately, the use of remote underwater video allows for easy detection of bias. It is possible to minimise the risk of observer bias via the use of adequate training and support for species identification. Future study on observer bias would benefit from a larger sample pool, possibly comparing observers of varying experience levels, as well as the effect of observer training. Future study with a broader data set is also recommended to determine the full extent of the effects that camera position has on monitoring the fish fauna of artificial reefs.
Chapter 5: Analysis of a cost-effective artificial reef monitoring method

5.1 Introduction

An essential component in assessing the biological performance of an artificial reef is the design of a robust monitoring program which can accurately detect changes in the abundance and diversity of fish fauna through space and time (Holmes et al. 2013). A wide variety of methods have been used to monitor marine communities in the past and the chosen technique should be based on the type of information required, the specific indices that need to be measured, the repeatability of the method, the level of precision required to detect change, as well as the environmental conditions in which monitoring will take place (Willis and Babcock 2000, Smale et al. 2011). The available time and financial resources to collect data must also be considered, as this can vary significantly depending on the selected monitoring regime (Langlois et al. 2010).

A frequent stumbling block encountered in many monitoring programs is the collection of sufficient data over large temporal and spatial scales when resources are limited (Baird et al. 2000). One solution to this is the use of volunteers to collect information. The use of volunteers, referred to as “citizen science”, to collect biological data is well established in both marine and terrestrial environments (Viswanathan et al. 2004, Wiber et al. 2004, Conrad and Daoust 2008, Conrad and Hilchey 2011, Gollan et al. 2012). The benefit of citizen science is that it allows a portion of monitoring costs to be borne by the volunteers, and has shown to increase stewardship of the resource (Pattengill-Semmens and Semmens 2003). However, with all volunteer based projects, monitoring regimes need to be developed that are both simple and effective, to ensure reliable data collection (Harding et al. 2000).

Recfishwest is currently involved in the development of a citizen science project aimed at using recreational fishers to collect information on the fish fauna of the Geographe Bay artificial reefs using underwater video monitoring. The goal of this project is the development of a cost-effective alternative to the use of dedicated researchers to carry out long-term biological monitoring of the artificial reefs.
Initial trials by Recfishwest involved the use of rotating remote underwater cameras, which provided a live feed of the video footage being collected to avoid collision with reef modules whilst monitoring. Analysis of the footage collected in these initial trials, however, showed this technique to be ineffective at monitoring the fish fauna of the artificial reefs. This was a result of the low quality of the footage collected by the cameras, and to high amounts of camera movement in rough weather while being suspended from the boat, both leading to an inability to distinguish between species (Florisson 2015). This led to a decision by Recfishwest, with the aid of Ecotone consulting, to trial the use of Baited Remote Underwater Video (BRUV) systems constructed from low cost materials.

This aim of this chapter was to investigate the types of information that can be extracted on the fish fauna of the Dunsborough and Bunbury artificial reefs by analyzing BRUV footage. This data was used to assess the ability of this method for monitoring the fish fauna on the reefs and how the fish assemblages on the Dunsborough and Bunbury artificial reefs varied.
5.2 Materials and Methods

5.2.1 Study Site

5.2.1.1 Geographe Bay

Geographe Bay is the southern-most protected marine embayment in south-west Australia and covers an area of roughly 290 square nautical miles. The bay has a maximum depth of 30 m and normally experiences a semidiurnal tide, with tidal movements averaging 0.5 m (Bellchambers et al. 2006). A more detailed description of Geographe Bay can be found in Section 4.2.

5.2.1.2 Geographe Bay artificial reefs

The Dunsborough reef is located at 115° 9.980’ E, 33° 33.962’ S, in 27 m of water, and the Bunbury reef is located at 115° 35.900’ E, 33° 18.500’ S at a depth of 17 m (Department of Fisheries Western Australia 2015). Each artificial reef consists of 30 ten-tonne reinforced concrete ‘Fish Box’ modules, placed in clusters of five, covering an area of roughly four hectares (Department of Fisheries Western Australia 2015). A more detailed description of the Geographe Bay artificial reefs can be found in Section 4.2.

5.2.2 Source of data

BRUV footage of the Dunsborough and Bunbury artificial reefs was collected from three separate sampling trips. Data collection took place on the 10\textsuperscript{th} and 19\textsuperscript{th} of March 2015 at the Dunsborough reef and the 25\textsuperscript{th} of May 2015 at the Bunbury reef. Data collection, the design of the sampling regime and the construction of the BRUV was performed solely by Ecotone consulting and Recfishwest, with no input from staff or students from Murdoch University.

During the final stages of this thesis, a preliminary species list was provided by the Western Australian Department of Fisheries (DoF), who have been monitoring the artificial reefs using a combination of Diver Operated Video (DOV) and BRUV since
the deployment of the reefs in 2013 (see appendix Table A5.1). The species list provided by the DoF contains a preliminary list of species that have been identified from six separate monitoring surveys of the Geographe Bay artificial reefs. Due to the short notice in which this information was obtained, it has not been included within the analysis of the results, however it has been used as comparative data set to assess whether the trends observed in the footage collected by Recfishwest and Ecotone consulting, are mirrored by that of a broader data set.

5.2.3 Sampling regime

The sampling method performed by Ecotone consulting and Recfishwest involved the haphazard dropping of BRUVs in the vicinity of the artificial reef modules using GPS for navigation. Each drop involved positioning the boat above a reef module and lowering the BRUV over the boat until it reached the sea floor. Camera submersion times averaged ~20 minutes, and upon retrieval, the video footage was extracted from the camera. The BRUV was then reset and rebaited before being deployed at a new location (Florisson 2015).

5.2.4 BRUV design

The frame of the BRUV used in this study was constructed from class 9 Polyvinyl Chloride (PVC) irrigation pipe. The frame of the BRUV was stabilised by two skids, each filled with four 680g lead weights, making the BRUV negatively buoyant. The bait used in this study was 500g of Sardine (Sardinops spp.), which was enclosed within a plastic mesh bait bag. The camera used to capture video footage was a GoPro Hero 4 Silver Action Video Camera™, which was placed in a waterproof housing. A more detailed description of the BRUV design can be found in Section 4.2.
5.2.5 Video analysis

Prior to analysis, the provided raw videos were coded according to their trip collection date (t), camera number (c), and video data number. For example a video collected on trip one, by camera one, with a video data number of 0001, would be coded (t1c1-0001). Two additional factors were given to each video that indicated the ‘reef’ that the footage was collected from and the camera ‘position’ as either facing reefs modules (F) or not facing reef modules (NF). The reason for including camera position as a factor in this study is due to previous work by Florisson (2015) and the findings of Chapter 4 of this thesis, which identified significant differences between the faunal compositions on footage collected from BRUVs facing towards reef modules and those facing away.

Thirty-three videos were analysed in total, with 24 from Dunsborough (12 facing reef modules, 12 not facing reef modules), and 9 from Bunbury (5 facing reef modules, 4 not facing reef modules). Each video was viewed for a 10-minute period between 7 and 17 minutes, giving a total of 330 minutes. Analysis of each video involved identifying each fish to the lowest possible taxonomic level, usually species, with the exception of Pseudocaranx spp., which require detailed examination (i.e. scale counts) to confidently distinguish between Pseudocaranx dentex and Pseudocaranx wrightii (Smith-Vaniz and Jelks 2006). An index of relative abundance (MaxN) was also recorded for each individual species. MaxN is defined as the maximum number of individuals of each species observed in a single frame over the sample period. MaxN is a widely used index in underwater video studies and provides a conservative measure of relative abundance that eliminates the chance of double counting (Willis and Babcock 2000, Cappo et al. 2003, Watson 2006). Whilst is not classified as a fish, Sepioteuthis australis (Southern Calamari), has been included within this study as it is an important recreational species with the Geographe Bay area and heavily targeted by fishers.

It has been noted that recommended soak time for BRUVs varies between 30 and 60 minutes in order to detect the majority of target species (Watson 2006, Watson et al. 2010, De Vos et al. 2014). However, this study was limited by the length of the videos collected and could only allow for a 7-minute bait soak time followed by a 10-
minute analysis of the footage. All video footage was reviewed by the author on an Apple Macintosh laptop computer using the multimedia program QuickTime.

Abundance data from each video were compiled into a single data matrix where each video had a unique identifier code as well as additional factors that indicted the reef that the footage was collected and the camera direction. All following statistical analysis was performed from this single data matrix.

5.2.6 Statistical analyses

All statistical analyses were undertaken using the Primer v7 multivariate statistics software package, with the PERMANOVA+ add on (Anderson et al. 2008, Clarke and Gorley 2015). In all analyses, the null hypothesis of no significant difference was rejected if the significance level ($p$) was ≤ 0.05.

5.2.6.1 Univariate analyses

Two-way Permutational Multivariate Analysis of Variance (PERMANOVA; Anderson et al., 2008) was employed to determine whether the values for taxon richness (number of taxa) and total MaxN (i.e. the sum of the MaxN values for each species in a sample) differed among sites (Bunbury and Dunsborough) and camera positions (facing towards and away from the artificial reef). Both of these variables were considered fixed. The DIVERSE routine was used to calculate, for each individual sample, the taxon richness and total MaxN.

Prior to subjecting the data for each dependent variable to two-way PERMANOVA, the extent of the linear relationship between the loge-transformed mean and loge-transformed standard deviation for each of the various sets of replicate samples for both variables was examined. This approach was used to determine whether the data for each variable required transformation to meet the test assumption of homogenous dispersions among a priori groups and, if so, to identify the appropriate transformation required (Clarke et al. 2014a). This analysis demonstrated that taxon
richness required a square root transformation, whilst total MaxN required a log(x+1) transformation.

The pre-treated data for each variable was then used to construct separate Euclidian distance matrices and subjected to the two-way PERMANOVA described above. Graphs of the transformed arithmetic means and associated ± 95% confidence intervals were plotted to visualise the extent of any differences among main effects.

### 5.2.6.2 Multivariate analysis

PERMANOVA, Analysis of Similarities (ANOSIM; Clarke and Green 1988) non-metric Multi-Dimensional Scaling (nMDS) ordination plots (Clarke 1993) and a shade plot (Clarke et al. 2014b, Tweedley et al. 2015) were employed to elucidate whether the composition of the fish and cephalopod faunas on the artificial reefs differed among sites and camera positions and, if so, the species that were responsible for those differences.

The MaxN for each species in each individual sample was subjected to a log(x+1) transformation to down weigh the contributions of highly abundant taxa and balance them with those of less abundant taxa. These transformed data were then used to construct a Bay-Curtis similarity matrix and subjected to the same two-way PERMANOVA test described above, only this time employing multivariate data. However, in this instance, the sole purpose of the PERMANOVA was to determine if there was an interaction between the site and camera position main effects and, if so, to determine the extent of those interactions relative to each other and to those of the main effects (Lek et al. 2011).

If the interaction was not significant, or relatively small in relation to the main effects, the matrix was then subjected to a two-way ANOSIM test. ANOSIM was preferred at this stage of the analysis because, unlike PERMANOVA, this test is fully non-parametric and thus more robust, and because the ANOSIM $R$-statistic provides a universal measure of group separation to test for significant interactions between region and position (Lek et al. 2011). The magnitude of the $R$ statistic typically ranges between
1, when the compositions of the samples within each group are more similar to each other than to that of any of the samples from other groups, down to ~0, when within-group and between-group similarities do not differ (Clarke et al., 2015).

The same Bray-Curtis similarity matrix was subjected to nMDS to produce an ordination plot, which provided a visual representation of the trends in faunal composition among the main effects. Finally, the log(x+1) transformed MaxN data for each species in each sample was then standardized and subjected to the Shade plot routine. This produced a visual display of the abundance matrix of variables (transformed and standardized species counts) against samples (each video), where the white represents the absence of a taxa in a sample and the intensity of grey-scale shading is linearly proportional to ‘abundance’ (Clarke et al. 2014b).

The order of both the variables and samples were determined independently (i.e. the order of variables is not influenced by the order of samples and vice versa) by the results of separate a group-average hierarchal agglomerative cluster analyses employing resemblance matrices defined using Whittaker’s index of association (Whittaker 1952, Valesini et al. 2014). Species exhibiting similar patterns of abundance across the samples were thus clustered together on the resultant dendrogram (y axis of the shade plot), while the samples (displayed on the x axis) were ordered by similarities in their ‘species’ composition. Note that, for clarity, only those taxa that occurred in two of more of the samples (i.e. 24 out of 35 taxa) were included in the shade plot.
5.3 Results

5.3.1 Mean density of species at artificial reef locations

A total of 35 taxa, from 22 families, including 34 fish and 1 cephalopod, were identified on BRUV footage, with the majority of taxa identified to species level (97%). The only taxa that could not be identified to species from the footage were from the genus *Pseudocaranx*. The most specious families on the video footage were Labridae and Carangidae, which were represented by five and three taxa respectively.

Thirty-four of the 35 taxa identified were present on footage from the Dunsborough reef (Table 5.1). The most abundant taxa identified at the Dunsborough reef were *Pseudocaranx* spp., which represented ~48% of the total abundance. The following most abundant species were *Coris auricularis* and *Trachurus novaezelandiae*, which represented ~15% and ~8% respectively, of the total abundance. A total of 11 taxa were identified on footage from the Bunbury reef. The most abundant species found on this footage was *C. auricularis*, which accounted for ~39% of the total abundance, followed by *Parequula melbournensis* (~31%) and *Neatypus obliquus* (~14%). Neither *Pseudocaranx* spp. nor *T. novaezelandiae*, were identified on footage from the Bunbury reef, however both *P. melbournensis* and *C. auricularis* were seen in higher abundance on the Bunbury reef, with mean MaxNs of 3.89 and 4.89 respectively, compared to 1.88 and 4.88 at Dunsborough reef. Of the 35 identified taxa, 23 taxa were restricted to the footage from the Dunsborough reef, whilst only a single species, *Trygonoptera personata*, was restricted to the footage from the Bunbury reef.
Table 5.1: Species table showing the mean MaxN (X) and standard error (SE) of each of the 35 fish and cephalopod taxa recorded using BRUVs on the Dunsborough and Bunbury artificial reefs. For each taxon, a percentage contribution (%) and ranking by mean MaxN (R) was calculated. Abundant species *i.e.* those that contributed ≥ 5 % to abundance recorded by any observer are shaded in grey.

<table>
<thead>
<tr>
<th>Species</th>
<th>Family</th>
<th>Total</th>
<th>Dunsborough</th>
<th>Bunbury</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td></td>
<td>X</td>
<td>SE</td>
<td>%</td>
</tr>
<tr>
<td><em>Pseudocaranx</em> spp.</td>
<td>CARANGIDAE</td>
<td>14.06</td>
<td>1.99</td>
<td>47.94</td>
</tr>
<tr>
<td><em>Coris auricularis</em></td>
<td>LABRIDAE</td>
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<td>0.82</td>
<td>15.29</td>
</tr>
<tr>
<td><em>Trachurus novaezelandiae</em></td>
<td>CARANGIDAE</td>
<td>2.24</td>
<td>1.35</td>
<td>7.65</td>
</tr>
<tr>
<td><em>Parequula melbournensis</em></td>
<td>GERREIDAE</td>
<td>1.88</td>
<td>0.42</td>
<td>6.41</td>
</tr>
<tr>
<td><em>Neatypus obliquus</em></td>
<td>KYPHOSIDAE</td>
<td>1.67</td>
<td>0.47</td>
<td>5.68</td>
</tr>
<tr>
<td><em>Anoplocapros amygdaloides</em></td>
<td>OSTRACIIDAE</td>
<td>0.85</td>
<td>0.16</td>
<td>2.89</td>
</tr>
<tr>
<td><em>Seriola hippos</em></td>
<td>CARANGIDAE</td>
<td>0.61</td>
<td>0.11</td>
<td>2.07</td>
</tr>
<tr>
<td><em>Austrolobatus maculatus</em></td>
<td>LABRIDAE</td>
<td>0.48</td>
<td>0.16</td>
<td>1.65</td>
</tr>
<tr>
<td><em>Upeneichthys vlamgingii</em></td>
<td>MULLIDAE</td>
<td>0.36</td>
<td>0.18</td>
<td>1.24</td>
</tr>
<tr>
<td><em>Trygonorrhina fasciata</em></td>
<td>RHINOBATIDAE</td>
<td>0.30</td>
<td>0.09</td>
<td>1.03</td>
</tr>
<tr>
<td><em>Sepioteuthis australis</em></td>
<td>LOLIGINIDAE</td>
<td>0.27</td>
<td>0.24</td>
<td>0.93</td>
</tr>
<tr>
<td><em>Pempheris klunzingeri</em></td>
<td>PEMPHERIDAE</td>
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<td>0.19</td>
<td>0.83</td>
</tr>
<tr>
<td><em>Diodon nicthermerus</em></td>
<td>DIODONTIDAE</td>
<td>0.24</td>
<td>0.10</td>
<td>0.83</td>
</tr>
<tr>
<td><em>Chelmolops curiosus</em></td>
<td>CHAETODONTIDAE</td>
<td>0.24</td>
<td>0.11</td>
<td>0.83</td>
</tr>
<tr>
<td><em>Myliobatis australis</em></td>
<td>MYLIOBATIDAE</td>
<td>0.21</td>
<td>0.07</td>
<td>0.72</td>
</tr>
<tr>
<td><em>Parapercis haackei</em></td>
<td>PINGUIPEDIDAE</td>
<td>0.15</td>
<td>0.09</td>
<td>0.52</td>
</tr>
<tr>
<td><em>Dasyatis breviceudata</em></td>
<td>DASYATIDAE</td>
<td>0.15</td>
<td>0.06</td>
<td>0.52</td>
</tr>
<tr>
<td><em>Chyrosophyrs auratus</em></td>
<td>SPARIDAE</td>
<td>0.12</td>
<td>0.06</td>
<td>0.41</td>
</tr>
<tr>
<td><em>Glaucosoma hebraicum</em></td>
<td>GLAUCOSOMATIDAE</td>
<td>0.09</td>
<td>0.05</td>
<td>0.31</td>
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</tbody>
</table>
Table 5.1 continued: Species table showing the mean MaxN (X) and standard error (SE) of each of the 35 fish and cephalopod taxa recorded using BRUVs on the Dunsborough and Bunbury artificial reefs. For each taxon, a percentage contribution (%) and ranking by mean MaxN (R) was calculated. Abundant species *i.e.* those that contributed ≥ 5% to abundance recorded by any observer are shaded in grey.

<table>
<thead>
<tr>
<th>Species</th>
<th>Family</th>
<th>X</th>
<th>SE</th>
<th>%</th>
<th>R</th>
<th>X</th>
<th>SE</th>
<th>%</th>
<th>R</th>
<th>X</th>
<th>SE</th>
<th>%</th>
<th>R</th>
</tr>
</thead>
<tbody>
<tr>
<td>Cheilodactylus gibbosus</td>
<td>CHEILODACTYLIDAE</td>
<td>0.09</td>
<td>0.05</td>
<td>0.31</td>
<td>19</td>
<td>0.13</td>
<td>0.07</td>
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<td>19</td>
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<td>Pentaceropsis recurvirostris</td>
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<td>0.04</td>
<td>0.21</td>
<td>21</td>
<td>0.08</td>
<td>0.06</td>
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<td>Parapercis ramsayi</td>
<td>PINGUIPEDIDAE</td>
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<td>0.04</td>
<td>0.21</td>
<td>21</td>
<td>0.08</td>
<td>0.06</td>
<td>0.23</td>
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<td>0.04</td>
<td>0.12</td>
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<td>0.08</td>
<td>0.06</td>
<td>0.23</td>
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</tr>
<tr>
<td>Choerodon rubescens</td>
<td>LABRIDAE</td>
<td>0.03</td>
<td>0.03</td>
<td>0.10</td>
<td>25</td>
<td>0.04</td>
<td>0.04</td>
<td>0.12</td>
<td>24</td>
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<td></td>
</tr>
<tr>
<td>Chromis klunzingeri</td>
<td>POMACENTRIDA</td>
<td>0.03</td>
<td>0.03</td>
<td>0.10</td>
<td>25</td>
<td>0.04</td>
<td>0.04</td>
<td>0.12</td>
<td>24</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Parazanclistius hutchinsi</td>
<td>PENTACEROTIDAE</td>
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<td>0.03</td>
<td>0.10</td>
<td>25</td>
<td>0.04</td>
<td>0.04</td>
<td>0.12</td>
<td>24</td>
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<td></td>
</tr>
<tr>
<td>Aracana aurita</td>
<td>OSTRACIIDAE</td>
<td>0.03</td>
<td>0.03</td>
<td>0.10</td>
<td>25</td>
<td>0.04</td>
<td>0.04</td>
<td>0.12</td>
<td>24</td>
<td></td>
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</tr>
<tr>
<td>Eubalichthys mosaicus</td>
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<td>0.03</td>
<td>0.10</td>
<td>25</td>
<td>0.04</td>
<td>0.04</td>
<td>0.12</td>
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<tr>
<td>Tilodon sexfasciatus</td>
<td>KYPHOSIDAE</td>
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<td>0.03</td>
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<td>25</td>
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<td>0.04</td>
<td>0.12</td>
<td>24</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Lagocephalus sceleratus</td>
<td>TETRAODONTIDAE</td>
<td>0.03</td>
<td>0.03</td>
<td>0.10</td>
<td>25</td>
<td>0.04</td>
<td>0.04</td>
<td>0.12</td>
<td>24</td>
<td></td>
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<td></td>
</tr>
<tr>
<td>Trygonoptera mucosa</td>
<td>UROLOPHIDAE</td>
<td>0.03</td>
<td>0.03</td>
<td>0.10</td>
<td>25</td>
<td>0.04</td>
<td>0.04</td>
<td>0.12</td>
<td>24</td>
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<td></td>
</tr>
<tr>
<td>Trygonoptera personata</td>
<td>UROLOPHIDAE</td>
<td>0.03</td>
<td>0.03</td>
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<td>25</td>
<td>0.11</td>
<td>0.11</td>
<td>0.88</td>
<td>8</td>
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<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Suezichthys cyanolaemus</td>
<td>LABRIDAE</td>
<td>0.03</td>
<td>0.03</td>
<td>0.10</td>
<td>25</td>
<td>0.04</td>
<td>0.04</td>
<td>0.12</td>
<td>24</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Pseudolabrus biserialis</td>
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<td>0.03</td>
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<td>0.04</td>
<td>0.12</td>
<td>24</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

| Species                   |              | 35   |     | 34 | 11 |
| Mean MaxN                 |              | 29   |     | 36 | 13 |
| # Samples                 |              | 33   |     | 24 | 9  |
5.3.2 Species diversity

PERMANOVA demonstrated that number of species differed significantly between the footage from the two reefs (Table 5.2A; Fig. 5.1A), but not between footage from different camera positions (Table 5.2A; Fig. 5.1B), with no significant interaction between reef and position (Table 5.2A). The mean number of species identified on the Bunbury and Dunsborough reef footage was roughly three and seven. As for camera position the mean number of species identified on reef facing and not reef facing footage was roughly six and five, respectively.

Table 5.2: Mean squares (MS), Pseudo-F (pF) values and significance levels (P) for a two-way PERMANOVA test on (A) number of species between reef and camera position and (B) abundance (total MaxN) between reef and camera position.

<table>
<thead>
<tr>
<th>(A) Number of species</th>
<th>df</th>
<th>MS</th>
<th>pF</th>
<th>P</th>
</tr>
</thead>
<tbody>
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<td>Reef</td>
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<td>4.15</td>
<td>18.62</td>
<td>0.001</td>
</tr>
<tr>
<td>Position</td>
<td>1</td>
<td>0.163</td>
<td>0.73</td>
<td>0.396</td>
</tr>
<tr>
<td>Reef x Position</td>
<td>1</td>
<td>0.013</td>
<td>0.057</td>
<td>0.805</td>
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<tr>
<td>Residual</td>
<td>29</td>
<td>0.223</td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

<table>
<thead>
<tr>
<th>(B) Abundance</th>
<th>df</th>
<th>MS</th>
<th>pF</th>
<th>P</th>
</tr>
</thead>
<tbody>
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<td>Reef</td>
<td>1</td>
<td>8.21</td>
<td>37.16</td>
<td>0.001</td>
</tr>
<tr>
<td>Position</td>
<td>1</td>
<td>1.49</td>
<td>6.737</td>
<td>0.016</td>
</tr>
<tr>
<td>Reef x Position</td>
<td>1</td>
<td>0.919</td>
<td>4.16</td>
<td>0.051</td>
</tr>
<tr>
<td>Residual</td>
<td>29</td>
<td>0.221</td>
<td></td>
<td></td>
</tr>
</tbody>
</table>
Chapter 5

5.3.3 Overall abundance

As for overall density, PERMANOVA identified significant differences between footage from the two reefs (Table 5.2B; Fig. 5.2A), and camera position (Table 5.2B; Fig. 5.2B). However, it should be noted the error values for relative abundance by position were large. As with the mean number of species, there was no significant interaction between reef and position in regards to abundance of species (Table 5.2B).

Fig. 5.1. Mean number of species, square root transformed, recorded at (A) the Bunbury and Dunsborough artificial reefs, and (B) by video footage facing reef modules (F) and not facing reef modules (NF). Error bars represent 95% confidence intervals.
Chapter 5

Fig 5.2. Mean abundance (MaxN), log(x+1) transformed, of individuals recorded at (A) the Bunbury and Dunsborough artificial reefs, and (B) by video footage facing reef modules (F) and not facing reef modules (NF). Error bars represent 95% confidence intervals.

5.3.4 Multivariate analysis

ANOSIM showed that the composition of species differed significantly between footage from the two reefs (Global R = 0.867, p = 0.001), but not for camera position (Global R = 0.071, p = 0.114), with PERMANOVA showing no significant interaction between reef and position (p = 0.817). The nMDS ordination plot, derived from the log(x+1) transformation of densities from all species, show clearly identifiable differences between regions (Fig. 5.3A), whilst the differences between positions are less clearly observable (Fig. 5.3B).
Fig. 5.3. An nMDS constructed using the Bay-Curtis Similarity matrix, using log(x+1) transformed data of the MaxN for each species in each sample. (A) Plot has been coded for reef with Dunsborough samples indicated by red, and Bunbury samples indicated by blue. (B) Plot has been coded for position with facing (F) samples indicated by purple, and not facing (NF) samples indicated by green.

A shade plot showing the percentage contribution to overall abundance of species that occurred in two or more samples only, highlights trends in individual species between both reef and camera position (Fig. 5.4). *Parequula melbournensis*, *S. hippos* and *C. auricularis* were found to occur frequently in samples from both reefs and camera positions; however *S. hippos* was found in lower numbers.
Species such as *Anoplocapros amygdaloides* and *Pseudocaranx* spp. were found in high numbers of video samples from the Dunsborough reef, but relatively few at the Bunbury reef. *Trachurus novaezelandiae*, which was the third most abundant species at the Dunsborough site occurred only in three samples, however in very high numbers. The shade plot also shows that species such as *Pentaceropsis recurvirostris* were found only to occur in footage that was collected facing reef modules whilst others such as *Dasyatis brevicaudata* and *Trygonorrhina fasciata*, were far more abundant in footage not facing reef modules.

In regards to recreationally important fish species, whilst *S. hippos* was found in similar abundance regardless of the reef or camera position, *Glaucosoma hebraicum*, *Chrysophrys auratus* and *Pseudocaranx* spp. were only identified on footage collected from the Dunsborough artificial reef. *Chyrosophyrs Auratus* was also only identified on footage that was collected facing away from reef modules (Fig 5.4).
Fig. 5.4. Shade plot illustrating species that were identified in two or more samples. Data has been log(x+1) transformed and converted to percentage contribution for each sample. Cluster analysis has grouped species and individual video samples by their similarity. Darker shading represents a greater percentage contribution.
5.4 Discussion

A total of 330 minutes of BRUV footage was analysed from 33 separate videos to gather information on the diversity and abundance of fish species on the Dunsborough and Bunbury artificial reefs. This footage was opportunistically obtained as a preliminary assessment of the use of cost-effective BRUVs to monitor the fish assemblages of the artificial reefs in Geographe Bay.

Whilst this chapter has compared footage between the two artificial reefs and found significant differences in the fish fauna, the limited data and the fact that this study has not taken into account any temporal variation has meant that only assumptions can be made as to the cause of these differences. This is owing to difficulty in knowing whether or not the similarities and differences regarding the fish fauna on the footage is indicative of real variation between the two artificial reefs or owing to limitations of the data.

Data collected by the DoF as part of a monitoring program has provided a baseline of the species diversity that can be expected to be found on the artificial reefs. Whilst this study provides only a preliminary analysis of the diversity and abundance of species on the artificial reefs, it also offers an opportunity to assess what improvements can be made in future monitoring of the reefs using BRUVs and recreational fishers.

5.4.1 Trends in the data between reefs

Significant differences for both the species diversity and the overall abundance of species were identified between the footage from the two reefs, with the Dunsborough reef having a greater diversity and abundance of species. One of the most significant differences observed between the two reefs was the absence of *Pseudocaranx* spp. and *T. novaezelandiae* from the footage of the Bunbury reef. Whilst *T. novaezelandiae* was the third most abundant species found at the Dunsborough reef, it only occurred in three of the 24 samples, and it is possible that the species was missed by chance at the Bunbury reef due to the limited amount of footage collected. The high abundance of the
species at the Dunsborough reef is a result of it being a schooling species that generally appears in high numbers, giving it a high MaxN count despite only occurring in a small number of samples (Hutchins and Swainston 1986, Froese and Pauly 2015).

*Pseudocaranx* spp. on the other hand was found in every video sample at the Dunsborough reef and would likely have been captured had it been present on the Bunbury reef in similar abundance at the time of collecting the footage. As this species has been detected at both regions by previous monitoring (Table A5.1), the lack of *Pseudocaranx* spp. on the BRUV footage from the Bunbury reef is likely not due to an absence of the species but rather a lower abundance, and possibly may have been detected with additional sampling. This may also be the case for other recreational target species such as *G. hebraicum* and *C. auratus*, which were only detected at the Dunsborough reef in this study, but have been shown to occur at both reefs (Table A5.1).

A wide variety of design and environmental factors can affect the abundance and diversity of species on artificial reefs. As the two reefs are constructed from identical materials and number of modules and located only 50 km apart it is expected that they would provide similar amounts of shelter and experience similar environmental conditions. Isolation from nearby natural reefs, however, has shown to be a key factor in determining the abundance of fish on artificial reefs. Specifically, research has shown that artificial reefs located further away from natural reefs have a greater abundance and diversity of both juvenile and adult species (Walsh 1985, Belmaker et al. 2005). These findings have been attributed to a lower level of predation on more isolated reefs and thus a higher abundance of prey species, such as *T. novaezelandiae* and *Pseudocaranx* spp. (Belmaker et al. 2005, Froese and Pauly 2015).

Another significant difference observed between the two reefs was the overall diversity of species. Thirty-five species from 22 families were identified overall, with 34 of these species found at the Dunsborough reef and 11 found at the Bunbury reef. Monitoring by the DoF identified a total of 57 taxa from six monitoring surveys, 25 of which were not recorded on the footage collected by Recfishwest and Ecotone consulting (Table A5.1). Of the total number of species identified by the DoF, 44 and
38 were detected at the Dunsborough and Bunbury reefs, respectively, using a combination of both BRUVs and DOV, with 31 taxa identified at both reefs using only BRUVs (Table A5.1). This indicates that whilst sampling was fairly effective at the Dunsborough reef, the lack of footage collected from the Bunbury reef may not have provided an accurate representation of the species composition on the reef.

5.4.2 Trends in data between camera direction

In contrast to previous research done by Florisson (2015), no significant difference was detected between footage collected facing and not facing reef modules. This is highlighted by relatively abundant species such as *Pseudocaranx* spp., *P. melbournensis*, *C. auricularis* and *S. hippos*, which were found in similar frequencies in both facing and not facing footage. These species are all inquisitive and opportunistic feeders and would have been quickly drawn in by the bait as well as the action of other fish at the BRUV regardless of the position of the camera (Hutchins and Swainston 1986, Froese and Pauly 2015).

There were, however, a number of species that showed a distinct preference to a specific habitat. Cryptic species such as *P. recurvirostris*, which is known to be shy and hide among structure, was detected only in footage that was facing the reef modules (Hutchins and Swainston 1986). Ray species on the other hand such as *T. fasciata* and *D. brevicaudata*, were found to be far more abundant on the sand and seagrass on the outskirts of the reef modules. This is likely due to the feeding preference of these species which prey on items in the sand and do not seek the protection of structure (Hutchins and Swainston 1986, Froese and Pauly 2015). As these species were only found in small numbers however, their effect on the analysis of camera position would have been lessened by more abundant species such as *P. melbournensis*, *C. auricularis* and *Pseudocaranx* spp.
5.4.3 Recommendations for future study

One of the major factors likely to influence estimates of fish abundance and diversity is the length of time that the BRUV is positioned on the seafloor to record footage, known as the soak time (Gladstone et al. 2012, Harasti et al. 2015). Previous studies using BRUVs have generally employed soak times between 30-60 minutes with longer times recommended to attract more ‘delayed reaction’ species (Stobart et al. 2007, Gladstone et al. 2012, Harvey et al. 2013). Increasing the soak time of BRUVs does, however, add extra costs, as this increases the time need to collected samples and analyze footage.

Willis and Babcock (2000) recommend a BRUV soak time of at least 30 minutes as this provides reliable estimates of relative abundance without incurring extra costs that provide little or no benefit. A study using BRUVs to monitor fish communities in the Abrolhos Islands found that a minimum soak time of 36 minutes is needed to detect the majority of species, with 60 minutes recommend to capture numerous target species (Watson 2006). Future BRUV monitoring of the artificial reefs using recreational fishers should aim for a minimum soak time of 30 minutes, as this is likely to provide sufficient data on the fish communities of the artificial reefs as well as minimize sampling costs. Gathering data over a greater temporal scale would also be beneficial, as whilst the footage collected in this study may represent the faunal composition of the reefs on the day of sampling, it is not able to provide information on seasonal variation.

Although no significant difference was observed between the facing of the cameras in this study, it should be taken into account that there were a number of species that may potentially be missed or detected in lower abundances depending on the direction of the camera. Increasing the BRUV soak time may also aid in reducing the variation between facing and not facing footage as a larger bait plume will attract fish from a greater area and reduce the effects of camera facing. However, additional research is needed to determine how this factor will affect the data collected in the long term and future study should continue to take note of the camera facing.

Although monitoring by the DoF has not looked at the differences between facing and not facing footage, they have detected significant differences in species
composition and abundance on different clusters of reef modules (Paul Lewis; Department of Fisheries WA pers.com. 2015). Variation between the clusters may be caused by a range of differences in ocean currents and sedimentation levels between exposed and protected reef modules (Pais et al. 2007). Haphazard dropping of BRUVs has been successfully used in the past to monitor fish assemblages, but it limits the amount of spatial analysis that can be done (Cappo and Brown 1996, Westera et al. 2003). By modifying the deployment method to ensure each cluster of modules is sampled separately and assigning each sample with a cluster code depending on its location (i.e. North cluster, South-West cluster etc.), analysis of the variation between clusters can be done in much the same way this study has compared the fish assemblages of the two artificial reefs.

Lastly, as well as comparing the two artificial reefs with each other, comparisons with natural reefs within Geographe Bay would also provide a good measure of the effectiveness of the artificial reefs (Carr and Hixon 1997). As the artificial reefs were designed to attract target species for recreational fishing, it would be useful to collect data on how the abundance of these species on the artificial reefs compares to that of natural reefs and whether the high visitation levels the artificial reefs receive from fishers is affecting fish populations (Carr and Hixon 1997, Department of Fisheries Western Australia 2015).

Considering the limited amount of data collected, as well as the fact that footage was collected from only a single trip to the Bunbury reef, and two to the Dunsborough reef, the use of cost-effective BRUV sampling does show potential to provide a successful long-term monitoring project. A number of significant differences were identified between the two reefs, but no distinct conclusions can be drawn due to the lack of data. However, these findings do warrant further investigation, and continued improvements to the sampling regime as well as monitoring over an extended temporal scale will provide more sufficient data to draw conclusions from.
## 5.5 Appendix

Table A5.1: Fish species recorded by the Department of Fisheries on the Bunbury and Dunsborough Reefs in the six monitoring surveys up to October 2014. Sampling was conducted using both Diver Operated Video (DOV) and Baited Remote Underwater Video (BRUV). Species are categorized by the region they were detected as well as the monitoring method that detected them. Shaded species are those that were not detected on the BRUV footage collected by Recfishwest and Ecotone consulting.

<table>
<thead>
<tr>
<th>Species</th>
<th>Dunsborough</th>
<th>Bunbury</th>
</tr>
</thead>
<tbody>
<tr>
<td>Anoplocapros amygdaloides</td>
<td>BRUV / DOV</td>
<td>BRUV / DOV</td>
</tr>
<tr>
<td>Anoplocapros lenticularus</td>
<td>BRUV</td>
<td>DOV</td>
</tr>
<tr>
<td>Apogon victoriae</td>
<td>DOV</td>
<td>BRUV / DOV</td>
</tr>
<tr>
<td>Aptychotrema victentiana</td>
<td>BRUV</td>
<td></td>
</tr>
<tr>
<td>Arcana aurita</td>
<td>BRUV / DOV</td>
<td>BRUV / DOV</td>
</tr>
<tr>
<td>Achoerodus gouldii</td>
<td>BRUV</td>
<td></td>
</tr>
<tr>
<td>Aulohalaelurus labiosus</td>
<td>BRUV</td>
<td></td>
</tr>
<tr>
<td>Austrolabrus maculatus</td>
<td>BRUV / DOV</td>
<td>BRUV / DOV</td>
</tr>
<tr>
<td>Caesioscorpis theagenes</td>
<td>BRUV / DOV</td>
<td>DOV</td>
</tr>
<tr>
<td>Cheilodactylus gibbosus</td>
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<td>BRUV / DOV</td>
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<td>Chelmolops curiosus</td>
<td>BRUV / DOV</td>
<td>BRUV / DOV</td>
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<td>BRUV / DOV</td>
<td>BRUV / DOV</td>
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<td>DOV</td>
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<td>Eupetrichthys angustipes</td>
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<td>BRUV / DOV</td>
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<td>Glaucosoma hebraicum</td>
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<td>Halichoeres brownfieldii</td>
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<td>Helcogramma decurrens</td>
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<td>Heniochus acuminatus</td>
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<td>Hypoplectrodes nigroruber</td>
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</tr>
<tr>
<td>Meuschenia freycineti</td>
<td>BRUV</td>
<td></td>
</tr>
<tr>
<td>Mustelus antarcticus</td>
<td>BRUV</td>
<td></td>
</tr>
<tr>
<td>Myliobatus australis</td>
<td>BRUV</td>
<td>BRUV</td>
</tr>
<tr>
<td>Neatypus obliquus</td>
<td>BRUV / DOV</td>
<td>BRUV / DOV</td>
</tr>
<tr>
<td>Neosebastes pandus</td>
<td>BRUV</td>
<td></td>
</tr>
<tr>
<td>Notolabrus parilus</td>
<td>BRUV / DOV</td>
<td></td>
</tr>
<tr>
<td>Ophthalmolepis lineolatus</td>
<td>BRUV</td>
<td></td>
</tr>
<tr>
<td>Parapercis haackei</td>
<td>DOV</td>
<td>DOV</td>
</tr>
</tbody>
</table>
Table A5.1 continued: Fish species recorded by the Department of Fisheries on the Bunbury and Dunsborough Reefs in the six monitoring surveys up to October 2014. Sampling was conducted using both Diver Operated Video (DOV) and Baited Remote Underwater Video (BRUV). Species are categorized by the region they were detected as well as the monitoring method that detected them. Shaded species are those that were not detected on the BRUV footage collected by Recfishwest and Ecotone consulting.

<table>
<thead>
<tr>
<th>Species</th>
<th>Bunsborough</th>
<th>Bunbury</th>
</tr>
</thead>
<tbody>
<tr>
<td><em>Paraplotosus albilabris</em></td>
<td>BRUV</td>
<td></td>
</tr>
<tr>
<td><em>Parapriacanthus elongatus</em></td>
<td>BRUV / DOV</td>
<td></td>
</tr>
<tr>
<td><em>Parequula melbournensis</em></td>
<td>BRUV</td>
<td>BRUV / DOV</td>
</tr>
<tr>
<td><em>Paristiopterus gallipavo</em></td>
<td>BRUV / DOV</td>
<td>BRUV</td>
</tr>
<tr>
<td><em>Parma mccullochi</em></td>
<td>BRUV / DOV</td>
<td></td>
</tr>
<tr>
<td><em>Parupeneus crysopleuron</em></td>
<td>BRUV</td>
<td></td>
</tr>
<tr>
<td><em>Pentapodus viitae</em></td>
<td>BRUV</td>
<td></td>
</tr>
<tr>
<td><em>Pempheris klunzingeri</em></td>
<td>BRUV / DOV</td>
<td>BRUV / DOV</td>
</tr>
<tr>
<td><em>Platycephelus sp.</em></td>
<td>BRUV</td>
<td>BRUV</td>
</tr>
<tr>
<td><em>Platycephelus specular</em></td>
<td>BRUV</td>
<td>BRUV</td>
</tr>
<tr>
<td><em>Platycephelus longispinis</em></td>
<td>BRUV</td>
<td></td>
</tr>
<tr>
<td><em>Pseudocaranx sp.</em></td>
<td>BRUV / DOV</td>
<td>BRUV</td>
</tr>
<tr>
<td><em>Pseudocaranx dentex</em></td>
<td>BRUV</td>
<td></td>
</tr>
<tr>
<td><em>Pseudolabrus biserialis</em></td>
<td>BRUV</td>
<td></td>
</tr>
<tr>
<td><em>Pseudorhombus jenynsii</em></td>
<td>BRUV</td>
<td></td>
</tr>
<tr>
<td><em>Seriola hippos</em></td>
<td>BRUV / DOV</td>
<td>BRUV</td>
</tr>
<tr>
<td><em>Siganus sp.</em></td>
<td>BRUV / DOV</td>
<td>BRUV</td>
</tr>
<tr>
<td><em>Tilodon sexfasciatus</em></td>
<td>BRUV</td>
<td>BRUV</td>
</tr>
<tr>
<td><em>Trachinops noarlungae</em></td>
<td>BRUV / DOV</td>
<td></td>
</tr>
<tr>
<td><em>Trachurus novaezelandiae</em></td>
<td>BRUV</td>
<td></td>
</tr>
<tr>
<td><em>Trygonoptera personata</em></td>
<td>BRUV / DOV</td>
<td></td>
</tr>
<tr>
<td><em>Trygonorrhina fasciata</em></td>
<td>BRUV</td>
<td>BRUV</td>
</tr>
<tr>
<td><em>Upeneichthys vlamigii</em></td>
<td>BRUV / DOV</td>
<td></td>
</tr>
<tr>
<td><em>Urolophus sp.</em></td>
<td>BRUV</td>
<td></td>
</tr>
</tbody>
</table>

| Total no. of species         | 44 | 38 |
| Total no of species detected by BRUV | 31 | 31 |
Chapter 6: Conclusion

This thesis describes the results of research on the design and application of artificial reefs and an evaluation of the efficacy of cost-effective methods for monitoring their fish faunas. A literature review of Habitat Enhancement Structures (HES) around the world, focusing primarily on artificial reefs, investigated the various materials, designs and uses of these structures (Chapter 2). The results demonstrated that these structures have been utilized for a wide variety of purposes ranging from sediment stabilization and mitigation of illegal trawling to the provision of additional habitat for nurseries, aquaculture and commercial and recreational fishing.

In order to maximise the effectiveness of these structures a variety of factors need to be taken into consideration to ensure the selected materials and designs are suited to the purpose. Whilst over 3000 articles have been published using the key words “artificial reefs” and/or “habitat enhancement structure(s)”, limited guidelines are available for the various materials and designs that exist (Tweedley, unpublished). Thus the data derived was used to construct a heat map (Fig 2.9) to provide advice for the best application of various materials and designs. This provides information to allow project managers planning to undertake a HES project to easily identify which designs and materials will be most suited to their intended purpose and help guide the future development of HESs.

Having researched the various designs, a literature review was taken to look at how trends in artificial reef construction within Australia have changed over time (Chapter 3). It was found that within Australia the past 10 years has seen a clear shift in the designs and materials used in artificial reef construction. Purpose-built concrete modules have replaced materials of opportunity (i.e. tyres and scuttled vessels) as the most prevalent reef building material. Whilst these materials require an additional cost, they provide significant long-term benefits such as increased reef longevity, species-specific designs and reduced environmental impact. With the number of artificial reefs set to increase in coming years, continued research is needed to provide up to date
information on the use of these structures and their socio-economic performance, *e.g.* do they increase tourism and generate trade for local fishing stores.

The deployment of artificial reefs can require financial investments within the millions and it is therefore important to evaluate their effectiveness. In the case of reefs such as those deployed in Geographe Bay to attract target species and enhance recreational fishing, it is essential to monitor how the fish faunas associated with these structures change over space and time. The citizen science project being conducted by Recfishwest and Ecotone consulting offers a potential cost-effective method for monitoring the fish fauna of the Geographe Bay artificial reefs. The use of volunteers allows for gathering information over larger temporal and spatial scales than would otherwise be possible with limited resources. However, when using volunteers with limited experience it is important to ensure that the video data collected is reliable.

To investigate the impact of observer bias, Baited Remote underwater Video (BRUV) footage provided by Recfishwest and Ecotone consulting was analysed by having multiple observers collect information on the fish fauna present on the footage (Chapter 4). It was found that whilst observers recorded similar species diversity and abundance counts, significant differences were present between their records of species composition. This indicates that the use of observers with limited experience in logging data from underwater video footage may lead to significant variation in the data set due to observer bias. If university students are to be used as part of the Recfishwest monitoring project, it is recommended that participants should receive additional training, particularly in species identification, and go through an initial trial period where their results are compared to that of a more experienced observer until a minimum similarity of 90% is consistently recorded.

Statistical analysis of footage collected from the Bunbury and Dunsborough artificial reefs was done to identify what level of information could be obtained using a cost-effective BRUV sampling method (Chapter 5). Analysis of the data found that significant differences in the species composition were present between the two reefs, but that modifications to the sampling regime and a broader data set are needed to provide a more accurate comparison. It is recommend that future monitoring of the
artificial reefs by recreational fishers should incorporate a minimum BRUV soak time of 30 minutes to provide an accurate representation of the target fish communities. As the Recfishwest monitoring project progresses and additional footage is collected over a greater temporal scale, a broader data set than what was available for this study will be able to be utilized and allow for a more accurate assessment and comparison of the fish faunas of the artificial reefs. Incorporating the monitoring of nearby natural reefs would also be effective in providing a comparative data set to assess the effectiveness of the Geographe Bay artificial reefs. A major limitation of this component was the amount of video footage that could be obtained within the time frame. However, this analysis was able to identify a number of factors such as observer bias and camera position that warrant further investigation and require additional research to better understand their implication on future monitoring of the artificial reefs using BRUVs and recreational fishers.

In summary, this research has provided information about patterns of artificial reef usage globally and in Australia, which aims to assist in the future development of artificial reefs. It has also provided a series of recommendations for training observers to minimise the risk of observer bias in future monitoring of the Geographe Bay artificial reefs. Finally, it has shown that a monitoring approach based on footage collected by custom-made BRUV devices has potential to provide a cost-effective means for monitoring the fish fauna of the Geographe Bay artificial reefs.
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