Characteristics of the fish faunas of artificial reefs in Geographe Bay determined from video footage collected by recreational fishers

Submitted by

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Declaration

I declare that this thesis is my own account of my research and contains as its main content work which has not been previously submitted for a degree at any tertiary education institution.

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31st October 2016

Abstract

The number of artificial reef deployments around Australia has increased in recent years due to their popularity amongst recreational fishers. As these reefs modify the environment and its associated fauna, monitoring is required to ensure that any negative impacts to the surrounding area are assessed and minimised. Given this and the high cost of purpose-built artificial reefs, there is a need to develop cost-effective monitoring methods to determine their faunal composition. To address this need, this thesis reviewed methods for monitoring the faunas of artificial reefs and utilised the Baited Remote Underwater Video (BRUV) method to survey the fish faunas of two artificial reefs in Geographe Bay.

Fourteen fauna monitoring methods, in their application to artificial reefs, were critically evaluated against five criteria, *i.e.* deployment, accuracy, precision, time and cost. Not all methods were found to be applicable to the different types of artificial reefs, with the accuracy of each technique depending upon the scale at which monitoring occurs and the type of fauna being targeted. The fastest and cheapest techniques were those that either utilised only minimal equipment and/or did not require observers. Remotely operated underwater video, particularly BRUVs, were found to provide a relatively inexpensive and effective tool for monitoring fish communities of artificial reefs.

This finding supported the choice of the BRUV method, which was deployed through citizen science, to monitor the fish communities of the Bunbury and Dunsborough artificial reefs in Geographe Bay, south-western Australia, between October 2015 and July 2016. The resultant videos were analysed, using two-way ANOVA, to determine if the number of taxa, total MaxN, Simpson's Index, as well as the MaxN of several key recreational species, differed between reefs and over time, whilst PERMANOVA was utilised to identify whether the composition of the fish communities differed spatially and temporally. Most of the 60 taxa recorded were resident teleosts, however, nine species of elasmobranch were also recorded. In terms of the number of individuals, most were either pelagic or epibenthic and fed on zooplankton or zoobenthos. Significant differences were found among reefs in all variables, except Simpson's Index, with greater values typically being recorded on the Dunsborough reef. Monthly differences were detected for the number of taxa, total MaxN and the abundance of two recreationally important species, with greater values occurring mainly during summer. The greatest differences in the above univariate variables and fish community composition were always found for the reef factor, indicating that the location of the reefs to nearby habitat was predominantly responsible for shaping their associated fish communities. The lower, but still influential, temporal differences were influenced by seasonal changes in water temperature and oceanographic currents.

The data collected during this study demonstrate that BRUVs, deployed through citizen science, can be a useful and cost-effective tool for monitoring the fish faunas of artificial reefs.

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Chapter 1: General introduction

1.1. Thesis structure

This thesis has determined the characteristics of the fish fauna present on two purpose-built artificial reefs deployed in Geographe Bay, Western Australia, over a 10-month period between October 2015 and July 2016 inclusive. To enable a thorough assessment of the fish assemblage data recorded during the study, the advantages and limitations of a range of commonly used fauna monitoring methods, in their application to artificial reefs, were critically reviewed. The research component of this thesis utilised one of these monitoring methods, Baited Remote Underwater Video (BRUV), deployed using a citizen science program, to record video footage on the two artificial reefs in Geographe Bay. These data were then used to elucidate whether the characteristics of the fish fauna differed between the two reefs, as well as over time. This chapter provides an overview to the thesis, by outlining a brief background to artificial reefs, the faunas they support and the legislated need to monitor them, before describing the rationale and aims of the study.

1.2. Artificial reefs

Habitat Enhancement Structures (HESs), which constitute both artificial reefs and Fish Aggregation Devices (FADs), are defined as materials purposefully placed in aquatic environments, primarily to modify ecological processes (Seaman, 2008; Department of Fisheries, 2012). These structures have been utilised around the world for a number of purposes, such as to increase the localised yield of recreationally and commercially targeted marine organisms, to act as a deterrent for trawling activity, to prevent coastal erosion and to provide additional sites for surfing and recreational diving (Bohnsack and

Sutherland, 1985; Baine, 2001; Simon et al., 2011). Types of HESs are separated based upon their area of deployment, with FADs being used in pelagic zones, while artificial reefs are placed exclusively on the substrate of aquatic environments (Seaman et al., 2011). Therefore, artificial reefs can be defined as materials purposefully placed on the substrate of aquatic environments, designed to meet a number of goals (Seaman, 2008; Seaman et al., 2011; Department of Fisheries, 2012).

Artificial reefs can be separated into two main types; (i) reefs composed of materials of opportunity, such as stone, wood, tyres, offshore oil platforms and shipwrecks, and (ii) those that are purpose designed and built, typically constructed from reinforced concrete and steel (Pickering and Whitmarsh, 1997; Baine, 2001). This thesis will focus solely on purpose built artificial reefs, omitting those composed from materials of opportunity as they are not classified as ‘true artificial reefs’ in Western Australia, as outlined in ‘*Policy on Habitat Enhancement Structures in Western Australia*’ (Department of Fisheries, 2012). Purpose-built reefs can be placed into three broad categories; those deployed in (i) shallow water, (ii) deep water, and (iii) those designed to mimic seagrass. Shallow water artificial reefs are generally constructed from concrete and are placed between depths of 10-30 metres (m), primarily to provide habitat for recreationally important fish species and for use in aquaculture (Bateman, 2015; Fisheries Research and Development Corporation, 2015; Department of Fisheries, 2016). Deep water artificial reefs, consisting of large steel modules, are deployed between depths of 30-150 m and are used to attract pelagic and deep demersal fish species (Bateman, 2015). Reefs which mimic seagrass can be composed of a variety of materials and are used to reduce shoreline erosion and provide additional habitat for fish and other marine organisms (Shahbudin et al., 2011). The research component of this thesis focuses on shallow water artificial reefs, while the literature review evaluates the application of fauna monitoring methods to all the broad categories of purpose-built artificial reefs.

1.3. Use of artificial reefs by fish

Despite the widespread deployment of artificial reefs, often with the stated purpose of providing habitat for recreationally important fish species, the ways in which fish utilise these habitats remain largely unknown. Considerable debate exists over whether artificial reefs increase fish abundance through attraction, or by the production of new individuals. The attraction hypothesis asserts that fish are drawn to artificial reefs due to behavioural preferences, but do not increase the carrying capacity or biomass of fish in the surrounding environment (Bohnsack, 1989; Brickhill et al., 2005). Alternatively, the production hypothesis postulates that artificial reefs are able to increase the carrying capacity of the environment and biomass of fish in the area. This is believed to be due to a greater number of juveniles surviving to adulthood, as a result of additional feeding and sheltering opportunities provided by increased structural habitat (Bohnsack, 1989; Grossman et al., 1997; Pickering and Whitmarsh, 1997; Pickering et al., 1999; Brickhill et al., 2005; Lowry et al., 2014).

Until relatively recently, the production hypothesis was generally accepted by the scientific community and served as the rationale for most artificial reef deployments (Grossman et al., 1997; Brickhill et al., 2005). However, the primary assumption of this hypothesis, *i.e.* that reef fish are limited by the abundance of hard substrates (Bohnsack, 1989), has recently been challenged (Brickhill et al., 2005). It has instead been suggested that reef fish are not always limited by hard substrata and that, in some circumstances, recruitment variability acts as the predominant limiting factor (Mapstone and

Fowler, 1988). Therefore, it cannot always be assumed that the placement of artificial reefs will increase the production of fish, rather they have the potential to contribute towards overfishing by concentrating the distribution of fish and increasing their catchability (Bohnsack, 1989; Grossman et al., 1997; Brickhill et al., 2005).

Bohnsack (1989) has posited that the opposing theories of attraction and production are likely not mutually exclusive, rather they occur along a continuum. The degree to which an artificial reef attracts or produces fish is dependent upon a number of factors, including the characteristics of the surrounding habitat and the artificial reef, as well as the biology and behavioural preferences of different species (Bohnsack and Sutherland, 1985; Brickhill et al., 2005). Therefore, fish utilisation of artificial reefs is unlikely to be static and can be expected to change over different time scales according to abiotic factors, as induced by diurnal and seasonal changes, and biotic factors, as predicted by the ecological succession theory (McCook, 1994).

1.4. Monitoring artificial reefs

With the lack of scientific consensus over how fish utilise artificial reefs, and their potential to exacerbate overfishing, there is a need to monitor the biology of artificial reefs over the lifetime of their deployment. Yoccoz et al. (2001) defines monitoring as *“the process of gathering information about some system at different points in time for the purpose of assessing system state and drawing inferences about changes in state over time”*. Thus, monitoring can occur at the ecosystem, habitat, population or community level to measure a number of different variables including, but not limited to, species richness, diversity, abundance and biomass. The level at which monitoring occurs and the variables measured are determined by the specific objectives of the monitoring program (Katsanevakis et al., 2012), which are primarily conducted for scientific or environmental management purposes. The requirement for environmental and biological monitoring to inform management, is often codified into legislation at the National and State level. This is commonly required for projects that have the potential to negatively impact upon the environment, including the deployment of artificial reefs.

The *London Convention and Protocol on the Prevention of Marine Pollution by Dumping of Wastes and Other Matter*, to which Australia is a signatory (International;International Maritime Organization, 2016), and the *Environment Protection (Sea Dumping) Act 1981* (Commonwealth; Department of the Environment, 2016a), have outlined the need for monitoring artificial reefs, in regards to their appropriate site and material selection. The requirement for biological and environmental monitoring of artificial reefs in Western Australia is covered under the *Fish Resources Management Act 1994* (Western Australia) and the *Environmental Protection and Biodiversity Act 1999* (Commonwealth; Department of Fisheries, 2012; Department of the Environment, 2016b). Therefore, depending upon the specifics of the project, there may be a legislated need to conduct biological monitoring of artificial reef deployments.

1.5. Rationale

The past ten years has seen a surge in the deployment of purpose built artificial reefs around Australia (Diplock, 2010; Bateman, 2015). These structures have typically been deployed for the purpose of enhancing recreational fishing, with reefs being located close to major cities and generally within popular fishing regions (Bateman, 2015). A recent deployment, known as the South West Artificial Reefs Trial, which began in 2013, conducted by the Department of Fisheries (Western Australia) and Recfishwest, involved the placement of artificial reef modules off Bunbury and Dunsborough in Geographe Bay, Western Australia (Government of Western Australia, 2013). As part of this trial, the Department of Fisheries (Western Australia) has undertaken an extensive monitoring program to ascertain the effect the reefs have had upon the surrounding biotic environment. With further artificial reef deployments scheduled, and already underway, around the state, *e.g.* off Mandurah, there is a need for the development of cost-effective methods to monitor their fauna (Florisson 2016, pers. comm., 5 May; Department of Fisheries, 2016; Recfishwest, 2016). This need was identified by Recfishwest and is currently being investigated, through funding provided by the Fisheries Research and Development Corporation Australia (FRDC-project number 2014/005), via a number of postgraduate research projects at Murdoch University. In part to meet funding requirements from the FRDC, a literature review critically evaluating the faunal monitoring methods available for artificial reefs was completed, which in turn helped to inform the evaluation of the BRUV method utilised in the research project.

As part of this research, Recfishwest has highlighted the potential of using citizen science to monitor the fauna of artificial reefs, as the technique, which uses volunteers, allows studies to overcome the expense and logistical difficulties associated with traditional marine monitoring (Cigliano et al., 2015; Hyder et al., 2015; Edgar et al., 2016). As a result of this, Florisson (2015) trialled the use of recreational fishers, as citizen scientists, to collect video footage on the fish fauna of the Bunbury and Dunsborough artificial reefs using live-action cameras. While Florisson (2015) was unsuccessful in collecting large quantities of data, the potential of using citizen science as part of a monitoring program was recognised, with the author making a suite of recommendations to develop the program. Following this research, Bateman

(2015) investigated the use of BRUVs as a means of monitoring the fish faunas of the artificial reefs of Geographe Bay, finding it to be both effective and low cost. However, this survey was only conducted over a small time frame resulting in limited data. As a result of these preliminary studies, this thesis has adopted the use of BRUVs, deployed through citizen science, to conduct a monitoring program of the fish communities of the artificial reefs of Geographe Bay, otherwise known as Reef Vision, over a significant time scale.

1.6. Aims

Given the need to develop cost-effective methods for monitoring the faunas of artificial reefs, the overall aim of this thesis was to build upon the previous work by Florisson (2015) and Bateman (2015) to determine the characteristics of the fish fauna present on the Bunbury and Dunsborough artificial reefs, using BRUVs deployed by recreational fishers. Specifically, the thesis has two main aims:

1. Conduct a critical analysis of the methods for monitoring the faunas of artificial reefs through a literature review (Chapter 2).

2. Determine the characteristics of the fish fauna present on two artificial reefs deployed in Geographe Bay, Western Australia, over a 10-month period between October 2015 and July 2016 inclusive (Chapter 3).

This thesis will provide; (i) a greater understanding of how fish utilise the artificial reefs of Geographe Bay, including determining whether the characteristics, and composition, of the fish fauna change throughout the year, and (ii) an evaluation of the utility of the monitoring program, which utilised

BRUV deployed through citizen science, indicating whether this method could be employed on future artificial reef deployments.

Chapter 2: A critical analysis of the methods for monitoring the faunas of artificial reefs

2.1. Abstract

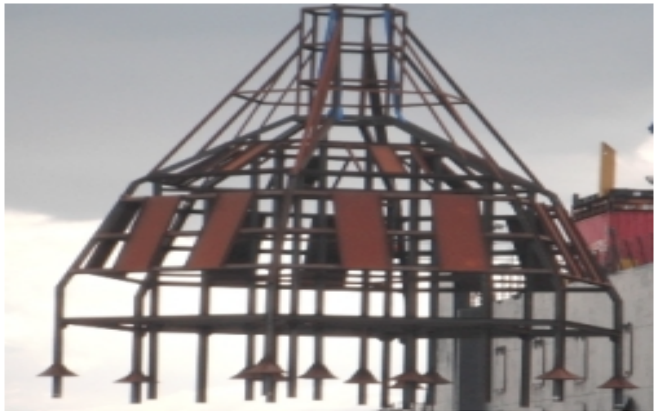
With the increasing number of artificial reef deployments in Australia in recent years, it is essential that appropriate monitoring regimes are in place to assess their impact upon surrounding biota. An integral component of any monitoring regime is the selection of an appropriate sampling method. In this review, I evaluated 14 methods for monitoring the faunas of artificial reefs against five criteria *ie.* deployment, accuracy, precision, time and cost. This review found that not all methods can be applied effectively to all types of artificial reefs and that the accuracy of the method depends upon the scale at which it operates, as well as the type of fauna being targeted for monitoring. Furthermore, underwater visual techniques, which employ minimal equipment, and methods that do not require the deployment of observers, are the fastest and cheapest methods to utilise. Therefore, as each technique has different advantages and limitations, monitoring methods should be evaluated and utilised according to the key questions and logistical circumstances of each study, rather than applying a one size fits all approach.

2.2. Introduction

Habitat Enhancement Structures (HES) are structures or materials placed in aquatic environments for the purpose of modifying ecological processes. The most common form of HES are artificial reefs, which are defined, in the Western Australian context, as purpose-built structures deployed exclusively on the substrate of aquatic environments (Seaman, 2008; Department of Fisheries, 2012). Under this definition artificial reefs can be placed into three broad categories; shallow-water, deep-water and artificial seagrass meadows

(Table 2.1; Bateman, 2015), with each type being deployed in different scenarios. Artificial reefs are used to meet a range of goals including to assist in environmental conservation and to provide social utility, however, they are used primarily to enhance fisheries (Bohnsack and Sutherland, 1985; Baine, 2001). This is achieved by increasing the localised abundance of fish via their attraction and concentration, or by improving their production through the provision of additional habitat, thereby inflating the carrying capacity of the natural environment (Bohnsack and Sutherland, 1985; Bohnsack, 1989; Seaman et al., 2011).

Table 2.1. Photographs of an example from each of the three broad categories of artificial reefs. Adapted from Bateman (2015).

Shallow-water artificial reefs		**Source**
		(left) Fish Box module. Fig. 2 in Haejoo (2016), (right) Reef Ball. Fig. 1 in Reef Ball Foundation (2016)
Deep-water artificial reef	**Artificial seagrass meadow**	
		(left) Wild Banks artificial reef. Fig. 1 in Department of National Parks Sport and Racing (2016), (right) Artificial seagrass trial. Fig. 1 in NIWA (2007)

The deployment of artificial reefs is covered under a variety of legislation at the International, Commonwealth and State levels. Under International law they are covered by the *London Convention and Protocol on the Prevention of Marine Pollution by Dumping of Wastes and Other Matter* (International Maritime Organization, 2016), to which Australia is a signatory. This agreement has been ratified into Australian Commonwealth legislation under the *Environment Protection (Sea Dumping) Act 1981*, which ensures appropriate site and material selection to minimise adverse impacts upon the environment and public (Department of the Environment, 2016a). Additional requirements may need to be met if the project has the potential to impact upon matters of national environmental significance, as defined under the Commonwealth legislation of the *Environmental Protection and Biodiversity Conservation Act 1999* (Department of the Environment, 2016b). In the state of Western Australia, the *Fish Resource Management Act 1994*, requires artificial reefs to be approved by the Department of Fisheries before deployment can occur, potentially requiring them to be monitored to ensure any negative impact upon the environment are assessed and minimised (Department of Fisheries, 2012). Therefore, a range of legislation requires that artificial reef deployments be monitored for a variety of reasons, including the need to assess their impact upon surrounding biota.

A plethora of different techniques are available for monitoring the faunas of natural and artificial reefs, however, there is no one ‘perfect’ method, as each one possesses its own inherent strengths, weaknesses and biases (English et al., 1997; Kingsford and Battershill, 2000; Hill and Wilkinson, 2004). Monitoring has been defined as *“the process of gathering information about some system at different points in time for the purpose of assessing system state and drawing inferences about changes in state over time”* (Yoccoz et al., 2001). As a result, each monitoring program must have specific objectives, informed by questions and/or hypotheses, which shape its design (Green, 1979; Bakus, 2007; Katsanevakis et al., 2012). It is these considerations that drive the level of investigation, from the ecosystem level to species specific studies, through the measurement of a range of variables, such as presence/absence, diversity, abundance and/or biomass (Katsanevakis et al., 2012). After the objectives of the study are defined, along with which variables will be measured, an appropriate monitoring technique or techniques must be chosen. Considerations towards sampling frequency and the type of statistical analysis to be used must be made at this stage, which will impact upon the monitoring method chosen. However, the logistical realities of each method, as well as the projects budget and time frame, must be considered. Thus, to ensure sound data are collected, care must be taken in selecting an appropriate method based upon the type of artificial reef being monitored, the purpose of the monitoring regime and the logistics of the study.

With the recent increase in deployment of artificial reefs in Australia, as well as the legislated requirements to monitor their impact on the surrounding environment, there is a need to critically analyse the suite of methods available for monitoring the fauna of these structures. As such, this review has critically analysed 14 monitoring methods, however, this does not represent an exhaustive list; rather these techniques are the ones most likely to be applicable for use with artificial reefs. The majority of these methods have been selected based on their use in coral reef and temperate rocky reef studies, as artificial and natural reefs share a number of biotic and abiotic attributes, *e.g.* habitat heterogeneity. Therefore, the advantages and limitations of these methods will likely be common between both habitats and allows for their inclusion in this review (Seaman, 2000). Methods drawn from other fields of research, such as fisheries assessments and emerging fields (*e.g.* environmental DNA analysis) have also been included due to their applicability across a wide range of habitats.

A total of 14 sampling methods (Table 2.2) were evaluated against five criteria, including:

- **Deployment:** Considerations for the ease of deployment and use of the method. This includes taking into consideration logistical issues such as transporting associated equipment, as well as the technical expertise and environmental conditions required to undertake the method.
- **Accuracy:** How close the method's estimates are likely to the 'true' population value, for both species richness and abundance. 'True' population values are extremely difficult to quantify, however, the assumption will be made that methods which provide higher estimates are more accurate. This assumption has been made as the majority of these methods employ visual observations, which are noted to under sample populations (Samoilys and Carlos, 2000). Thus, higher observations are likely to produce a value closer to the 'true' population value.
- **Precision:** The level of variation, between samples, of the population estimates. Lower variation is desired as this allows for the easier detection of change (Andrew and Mapstone, 1987; Samoilys and Carlos, 2000).
- **Time:** The overall time required to undertake monitoring, and analyse the results.
- **Cost:** The overall cost of method-specific equipment and the expense of undertaking monitoring and laboratory based analysis. The expense of undertaking monitoring and analysis is closely linked to time, as the longer the duration, the greater the cost, though laboratory work time is generally less expensive than field work time.

Considerations were also given to the effectiveness of each method for use with monitoring the three broad categories of artificial reefs (deep-water, shallow-water and artificial seagrass), their ability to undertake monitoring over a small (fine scale), moderate (medium scale) and wide areas (broad scale), as well as the methods ability to provide quantitative data.

Each method was given a subjective score out of 100, for each criterion, based upon the individualised summary tables. Scores of 0-20 will be designated as ‘very ineffective’, 20-40 as ‘ineffective’, 40-60 as ‘moderately effective’, 60-80 as ‘effective’ and 80-100 as ‘very effective’. This information will then be summarised in a ‘heat map’, providing a graphical representation of the qualitative data from each method in a matrix through the shading of tiles according to a colour scale (Wilkinson and Friendly, 2009). This scale ranged from red (very ineffective) to dark green (very effective) (Fig. 2.8). From the ‘heat map’ conclusions can be drawn as to the effectiveness of each method against the criteria, allowing for the observation of general trends.

Table 2.2. A summary of the fauna monitoring methods evaluated in this review.

Categories of fauna and their monitoring methods	Variations
Sessile/sedentary fauna	
Settlement tiles	Direct attachment Raised racks
Visual quadrats	Transect Random
Photo quadrats	Transect Random Photo point monitoring
Mobile fauna	
Stationary visual census	Nested sampling
Rapid visual technique	
DIDSON acoustic survey	
Sessile/sedentary and mobile fauna	
Visual transects	Point intercept Line intercept
Video transects	
Manta tow	
Towed video	Seabed tow Mid-water tow Towed diver video
Remotely operated underwater video	Linked Autonomous
Environmental DNA analysis	
Extractive methods	Fish trap Trawls Ichthyocide Hook and line
Fisher surveys	Onsite surveys Offsite surveys

2.2. Monitoring methods for sessile/sedentary fauna

2.2.1. Settlement tiles

Settlement tiles (or plates) are a monitoring method utilised in studying sessile benthic organisms, such as algae and invertebrates, and provide quantitative data on a fine spatial scale (Hill and Wilkinson, 2004; Muth, 2012; Janßen et al., 2013; Schloder et al., 2013; Guy-Haim et al., 2015). The method involves deploying a series of tiles, which can be attached directly to hard substratum and/or placed on raised racks (Fig 2.1; Mundy, 2000; Hill and Wilkinson, 2004). The tiles are later collected and analysed to determine the abundance and diversity of newly settled organisms (Muth, 2012; Ferse et al., 2013; Janßen et al., 2013; Guy-Haim et al., 2015). This method, in particular, has been used extensively in coral reef research (Field et al., 2007). In brief, as shown in Table 2.3 and Fig. 2.8, the settlement tile method was highlighted as providing relatively accurate species richness and abundance estimates, with a high level of precision, and could be utilised for monitoring sessile benthic organisms that settle directly onto the surface of artificial reefs. However, the deployment of the method can be difficult, time consuming and expensive.

Table 2.3. The advantages and limitations of the settlement tile method for monitoring sessile/sedentary fauna on artificial reefs.

Advantages	Limitations
Deployment	
• Settlement tiles can be deployed in a variety of locations, depending on the attachment method, allowing for site-specific studies (Field et al., 2007).	• The depth of tile deployment is limited by diver occupational health and safety. • The method is invasive as biotic samples are taken. • The equipment required for the deployment of tiles is cumbersome, regardless of the attachment method (Hill and Wilkinson, 2004). • Settlement plate material can bias settlement (Field et al., 2007). • The laboratory-based analysis requires considerable expertise to conduct species level identifications (English et al., 1997; Field et al., 2007).
Accuracy	
• The accuracy of the method has not been compared against other similar benthic monitoring methods. However, because the tiles are analysed in a laboratory it is assumed that it will allow for the accurate enumeration of organisms present (Kingsford and Battershill, 2000).	• Identifying settled taxa to species level can be difficult and require taxonomic expertise (Hill and Wilkinson, 2004). • Racks can alter the hydrodynamics of the water column and thus bias the settlement of organisms onto the tiles (Field et al., 2007).
Precision	
• The method has a high level of precision (Hill and Wilkinson, 2004).	
Time	
• Tiles, if mounted in racks, can be deployed quickly in the field (Field et al., 2007).	• Laboratory-based identification is time consuming (Hill and Wilkinson, 2004).
Cost	
	• The method is cited as being expensive, due to the high level of expertise and time required to identify the species present on the tiles (Hill and Wilkinson, 2004).

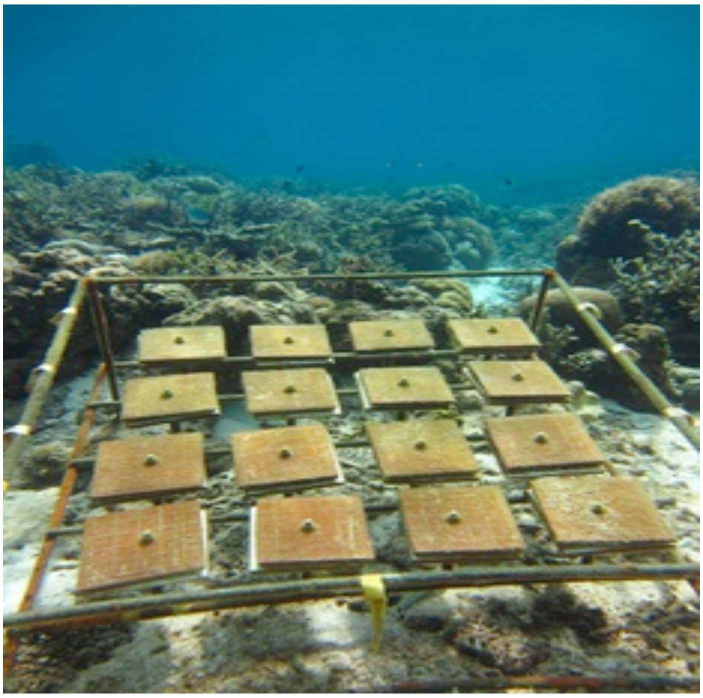


Fig. 2.1. Settlement tiles placed on raised racks (from Fig. 9 in Bremen (2014)).

2.2.2. Visual quadrats

The quadrat method was originally developed in the terrestrial plant ecology field (Beenaerts and Berghe, 2005), but has since been employed in aquatic environments. Deployment involves placing the quadrat(s) along transect lines or at random (Dodge et al., 1982; Miller and Ambrose, 2000), within a study area. The resultant quantitative data are usually based on estimates of the abundance or percentage cover of various organisms within the quadrat and provide information at fine spatial scales (Fig 2.2; Hill and Wilkinson, 2004; Beenaerts and Berghe, 2005). These estimates are primarily made by either direct visual observation (Lessios, 1996) or through point count methods (Fig. 2.2; Foster et al., 1991). The method is typically used for monitoring sessile and sedentary benthic communities and individual organisms (*i.e.* Taylor, 1998; Duarte and Kirkman, 2001; Pehlke and Bartsch, 2008; Parravicini et al., 2010; Mantelatto et al., 2013; Schonberg, 2015). Visual quadrats were shown to be a relatively easy method to undertake, which in conjunction with its accuracy in estimating species diversity and abundance, could be utilised to monitor sessile/sedentary organisms that live on, and immediately around the reef (Table 2.4, Fig. 2.8). Additionally, considerations must be made as to the time and financial investment required to undertake the method, as they can be considerable.

Table 2.4. The advantages and limitations of the visual quadrat method for monitoring sessile/sedentary fauna on artificial reefs.

Advantages	***Limitations***
Deployment	
• As observation takes place close to benthos, the method is not reliant on highly transparent water (Mantelatto et al., 2013). • Biological samples (specimens) can be taken *in situ* for further analysis and/or identification (Provost et al., 2013).	• Diver observers are limited in the duration and depth of their fieldwork by occupational health and safety. • Highly trained observers are required to collect the data (Mantelatto et al., 2013).
Accuracy	
• Visual quadrats typically capture cryptic species that may be missed in other methods (Obermeyer, 1998; Parravicini et al., 2010; Jokiel et al., 2015). • A number of studies have found visual quadrats to provide higher cover estimates for coral and benthic communities than photo quadrats, visual transects and video transects (Weinberg, 1981; Foster et al., 1991; Leujak and Ormond, 2007; Mantelatto et al., 2013). Though Jokiel et al. (2015) found the method to provide lower cover estimates for coral than both visual and video transects.	• Only organisms in the quadrat are sampled (*e.g.* 50 cm x 50 cm), so the method captures only a small area. • The method is likely to underestimate the abundance of species that occur in low densities due to the small area sampled (McClanahan and Muthiga, 1992; Leonard and Clark, 1993; Parravicini et al., 2010).
Precision	
	• Visual quadrats are regarded as having a moderate to high level of precision (Foster et al., 1991; Hill and Wilkinson, 2004; Jokiel et al., 2015).

Advantages	Limitations
Time	
• The method requires little, if any, laboratory time and has been described in multiple studies as being overall relatively time efficient (Foster et al., 1991; Leonard and Clark, 1993; Mantelatto et al., 2013; Jokiel et al., 2015).	• As all data is collected *in-situ,* fieldwork can be time intensive (Jokiel et al., 2015).
Cost	
• Quadrats are relatively inexpensive to produce (Leujak and Ormond, 2007; Jokiel et al., 2015).	• The level of fieldwork required can make the method costly to carry out, particularly so if SCUBA is involved (Mantelatto et al., 2013).

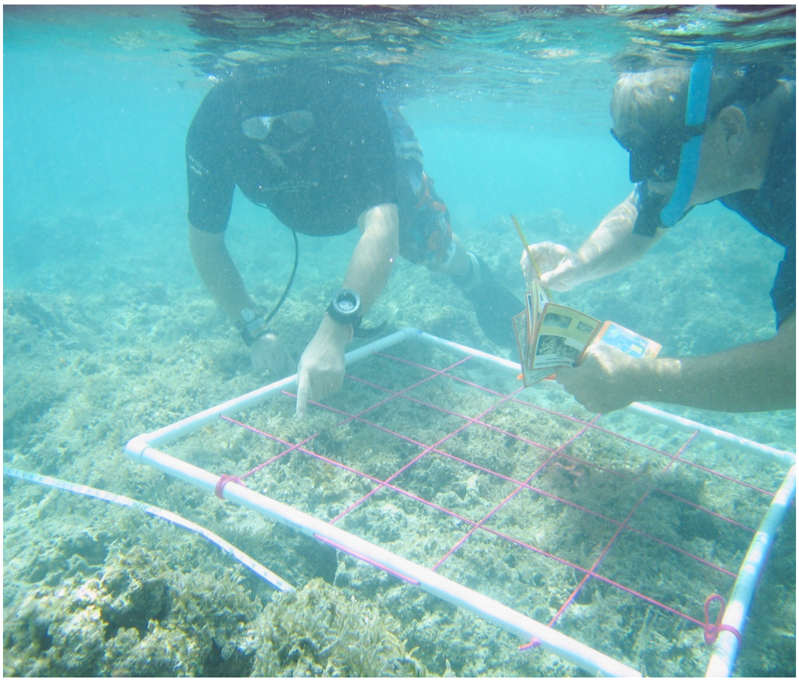


Fig. 2.2. Observers conducting a visual quadrat (from Fig. 1 in Reef Watch Waikiki (2010))

2.2.3. Photo quadrats

The photo quadrat method is a modification of the visual quadrat method where data is recorded from photographs, taken using a camera mounted on a frame, rather than by *in situ* visual census (Fig. 2.3; Mantelatto et al., 2013). It provides fine spatial scale quantitative data, similar to the visual quadrat method, by providing abundance and percentage cover estimates of benthic organisms (Carney et al.; Roberts and Davis, 1996; Hill and Wilkinson, 2004;

Kutser et al., 2007; Gilby et al., 2015). A variation of the photo quadrat method is photo point monitoring, where quadrats are permanently fixed to the substrate to allow for the monitoring of change over time at a fixed location (Lanyon and Marsh, 1995; Lessios, 1996; Bak et al., 2005). The photo quadrat method provides relatively accurate abundance estimates, whilst delivering effective precision between samples (Table 2.5, Fig. 2.8). The method can be deployed relatively easily and could be utilised for surveying sessile/sedentary organisms that live on, and immediately around, artificial reefs. Though its ability to detect accurate species richness estimates may be limited, and prove to be a costly and relatively time intensive undertaking.

Table 2.5. The advantages and limitations of the photo quadrat method for monitoring sessile/sedentary fauna on artificial reefs.

Advantages	***Limitations***
Deployment	
• Fieldwork can be conducted by researchers without taxonomic expertise (Mantelatto et al., 2013). • The resultant photograph is a permanent record of the data that can be reanalysed later if required (Mantelatto et al., 2013).	• The extent of fieldwork conducted by divers is limited by occupational health and safety. • The method requires a high level of water clarity, more so than visual quadrats (Mantelatto et al., 2013).
Accuracy	
	• Only samples a small area. • Obscuring can occur, leading to missed or unidentified organisms (Weinberg, 1981). • Multiple coral studies have found the method to record lower species richness than visual quadrats (Leujak and Ormond, 2007; Mantelatto et al., 2013; Jokiel et al., 2015), though Weinberg (1981) found it to capture higher estimates than visual quadrats. • Benthic community studies have found the method to record lower cover estimates than visual quadrats (Weinberg, 1981; Foster et al., 1991).

Advantages	Limitations
Precision	
• The method has been noted to provide a moderate to high level of precision for the monitoring of coral communities (Foster et al., 1991; Jokiel et al., 2015).	
Time and Cost	
• Photo quadrats require less time in the field than visual quadrats, which reduces costs (Mantelatto et al., 2013; Jokiel et al., 2015).	• Analysis of the species in the photographs can take considerable amounts of time (Mantelatto et al., 2013; Jokiel et al., 2015). • Photographic equipment (including underwater lights) can be expensive (Mantelatto et al., 2013; Jokiel et al., 2015).

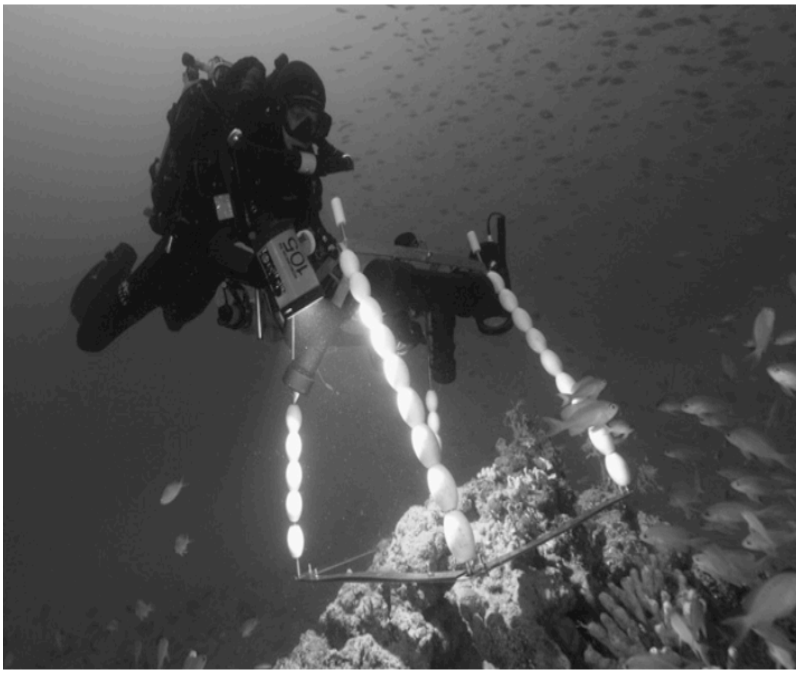


Fig. 2.3. A diver undertaking a photographic quadrat (from Fig. 3 from Deter et al. (2012)).

2.3. Monitoring methods for mobile fauna

2.3.1. Stationary visual census

The stationary visual census monitoring method was developed for studying reef associated fish (Bohnsack and Bannerot, 1986). The method involves a

SCUBA diver floating above a randomly selected point observing fish within an imaginary predefined cylinder (*i.e.* census area) around them (Cappo and Brown, 1996; Ayotte et al., 2015). This is typically done over a ten-minute period, which provides quantitative data on a medium to fine spatial scale (Bohnsack and Bannerot, 1986; Cappo and Brown, 1996; Hill and Wilkinson, 2004; Ayotte et al., 2015). The stationary visual census method is a technique which can be easily undertaken, in a short period of time, for low cost, though the accuracy of its species richness and abundance estimates may be limited (Fig. 2.8, Table 2.6). The method could be utilised for the monitoring of fish and/or mobile pelagic/demersal invertebrates (*i.e* cephalopods) utilising artificial reefs.

Table 2.6. The advantages and limitations of the stationary visual census method for monitoring mobile fauna on artificial reefs.

Advantages	***Limitations***
Deployment	
• Data collection is simple and requires minimal equipment (Hill and Wilkinson, 2004).	• The number of sampling events by divers is limited by occupational health and safety. • Field observers require considerable taxonomic skills.
Accuracy	
• The method avoids biases present in other underwater visual census techniques that utilise moving observers (*i.e* differences in swimming speeds) (Bohnsack and Bannerot, 1986) and induced behavioural responses of fish due to observer movement (Bohnsack and Bannerot, 1986; Colvocoresses and Acosta, 2007).	• The method only samples a small area (Colvocoresses and Acosta, 2007). • Subjectivity of the census area makes absolute densities unreliable (Hill and Wilkinson, 2004). • The method is likely to underestimate cryptic and demersal species (Riberio et al., 2004; Minte-Vera et al., 2008; Green et al., 2013). • Riberio et al. (2004) noted the method to have recorded significantly less species of reef fish than both visual transect and the rapid visual technique.

Advantages	Limitations
• Lowry et al. (2012) found the method to have recorded a higher species richness of estuarine fish than BRUV. However, it should be noted that the authors used a modified version of the method allowing the observer to move at the end of the initial period to note species not previously encountered.	• Both Colvocoresses and Acosta (2007) and Samoilys and Carlos (2000) found this method recorded lower abundances of reef fish compared to visual transects. • Interestingly, Minte-Vera et al. (2008) found that a smaller census area allowed for more accurate estimates of the abundances of small reef fish, and that increasing the census area allowed for more accurate estimates of large reef fish. This method, known as nested sampling, should be utilised to allow for the more accurate abundance estimations of both small and large fish.
Precision • The method has a moderate to high level of precision (Bohnsack and Bannerot, 1986; Hill and Wilkinson, 2004). *Time* • Samoilys and Carlos (2000) found the method to be faster, in terms of fieldwork, than visual transects. *Cost* • Stationary visual census surveys were found by Minte-Vera et al. (2008) to be cheaper than underwater visual transects.	

2.3.2. Rapid visual technique

The rapid visual technique has been used extensively for monitoring reef-associated fish (Sanderson and Solonsky, 1986; Kellison et al., 2012; Rizzari et al., 2014), providing medium to fine spatial scale qualitative data (Hill and Wilkinson, 2004). Data is collected by observers who swim randomly around a reef, at a constant depth and speed, for a set period of time searching for fish

(Hill and Wilkinson, 2004). Species are ranked according to when they are first observed, which allows for relative abundances to be calculated based upon the assumption that species encountered earlier are likely to be the most abundant (Jones and Thompson, 1978; Sanderson and Solonsky, 1986). The rapid visual technique, like the stationary visual census method, is a quick and cost-effective method, which can be carried out easily (Table 2.7, Fig. 2.8). Again, its species richness and abundance estimates must be treated with caution. It could be used to potentially monitor the abundance of fish and/or pelagic/demersal invertebrates (*i.e* cephalopods) utilising artificial reefs.

Table 2.7. The advantages and limitations of the rapid visual technique for monitoring mobile fauna on artificial reefs.

Advantages	***Limitations***
Deployment	
• The method can be used when conditions and/or habitats prevent the use of transects (Sanderson and Solonsky, 1986). • Minimal equipment is required for fieldwork (Hill and Wilkinson, 2004).	• Observers are limited in their dive range and time by occupational health and safety. • The use of field observers with considerable taxonomic skills is required.
Accuracy	
• The rapid visual technique has been shown to capture higher species richness and abundance than visual transects for reef fish (Kimmel, 1985) and greater abundance than remotely operated underwater video for reef sharks (Rizzari et al., 2014).	• The presence of a human observer may alter the behaviour of fish (Hill and Wilkinson, 2004). • The rapid visual technique is based on the assumption that species encountered earlier on in the sampling period are likely to be the most abundant (Jones and Thompson, 1978), which fails to account for differences in the spatial distribution of species (DeMartini and Roberts, 1982). This can lead to the overestimation of the abundance of widespread rare species and an underestimation of those with patchy but abundant distributions (DeMartini and Roberts, 1982).

Advantages	Limitations
	• This method is less able to detect cryptic species (Jones and Thompson, 1978). • The data generated is relative and does not provide quantitative abundances (densities) due to an unknown area being sampled (Kingsford and Battershill, 2000; Riberio et al., 2004; Taillon and Fox, 2004).
Precision • The method has a moderate level of precision (Sanderson and Solonsky, 1986; Hill and Wilkinson, 2004). *Time and Cost* • Data is collected relatively quickly and has low costs (Hill and Wilkinson, 2004), with Sanderson and Solonsky (1986) determining that this method is a useful substitute when time in the field and/or budget is limited.	

2.3.3. DIDSON acoustic survey

DIDSON (Dual frequency IDentification SONar) is an acoustic sounder that uses sonar to create video-like images (Fig. 2.4; Belcher et al., 2001; Moursund et al., 2003; Baumgartner et al., 2006; Martignac et al., 2015). The technology was originally developed for the United States Navy to detect underwater intruders in harbours (Belcher et al., 2001; Moursund et al., 2003) and has since been co-opted for ecological studies. The technology can be utilised to identify and quantify mobile fauna on a medium to large spatial scale (Kingsford and Battershill, 2000). It has been used extensively in monitoring fish migration, particularly at night (Galbreath and Barber, 2005; Baumgartner et al., 2006; Holmes et al., 2006; Pavlov et al., 2009), and fish assemblages in turbid and/or complex environments where visual or camera surveys would not provide reliable data (Frias-Torres and Luo, 2008; Becker et al., 2011; Crossman et al., 2011; Able et al., 2014). The DIDSON acoustic survey technique provides a quick and easy method to deploy. However, its usefulness may be limited by its expense, and issues inherent to the technology, which limit its species richness accuracy (Table 2.8, Fig. 2.4). In the right circumstances it could be utilised to record the abundance of fish and/or mobile pelagic/demersal invertebrates (*i.e.* cephalopods) utilising artificial reefs.

Table 2.8. The advantages and limitations of the DIDSON acoustic survey method for monitoring mobile fauna on artificial reefs.

Advantages	***Limitations***
Deployment	
• The method is non-invasive (Martignac et al., 2015). • The DIDSON unit is compact, lightweight and uses little power, allowing for easy deployment (Belcher et al., 2001; Moursund et al., 2003; Able et al., 2014). • The equipment can be employed in environments where it is too dark or turbid for other direct observation methods, or where extractive means are not practical (Belcher et al., 2001; Moursund et al., 2003; Baumgartner et al., 2006; Able et al., 2014).	• The deployment of the technology is currently limited to shallow depths (Galbreath and Barber, 2005), due to its relatively low detection range, 42 m (Belcher et al., 2001; Han and Uye, 2009).
Accuracy	
• Frias-Torres and Luo (2008) showed that DIDSON recorded a higher number of juvenile Goliath Grouper than remotely operated underwater video in a complex mangrove environment.	• The technology provides relatively low-resolution images. This can make it difficult to differentiate between species (Able et al., 2014), unless obvious morphological characteristics are present (Langkau et al., 2012), however, higher resolution models are in development.

Advantages	*Limitations*
	• The surrounding environment and the number of organisms present can impact the output image (Cronkite et al., 2006; Martignac et al., 2015). This likely contributes to its under sampling of small and benthic species which has been noted by Able et al. (2014) in estuarine habitats. • DIDSON was shown by Baumgartner et al. (2006) to mostly detect a lower abundance and species richness of migrating estuarine fish, than both standard cage trap and open-topped pop-nets.
Precision • The method has been noted as providing a high level of precision in species specific studies, but this is likely to be density dependent (Holmes et al., 2006).	• Precision is likely to be limited for studies covering multiple species, or in habitats with high diversity, as differences between observers could bias results.
Time • The technology provides real-time monitoring (Moursund et al., 2003). • The method can be used to sample large areas quickly (Kingsford and Battershill, 2000).	
Cost • The method has been cited by Martignac et al. (2015) as being of moderate cost, allowing for cost-effective monitoring.	

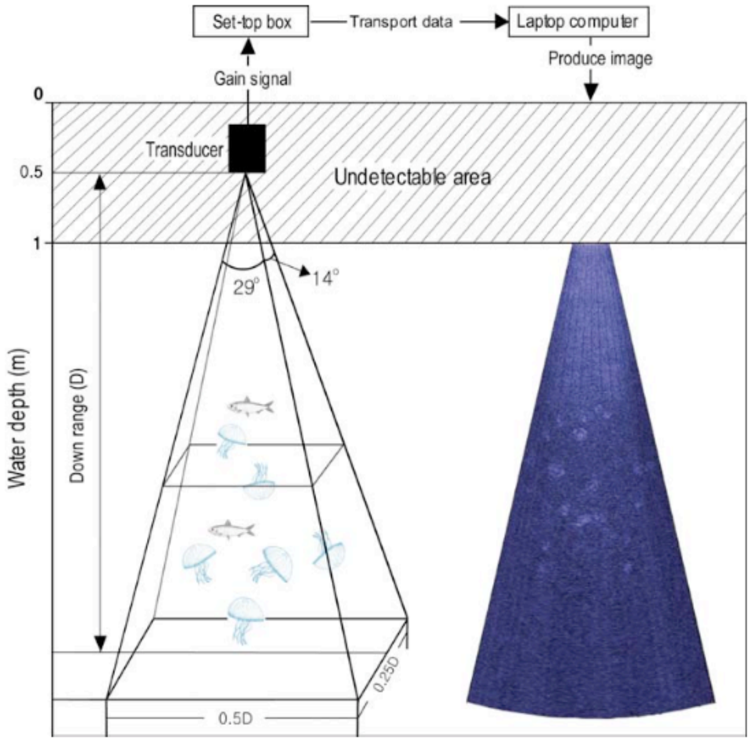


Fig. 2.4. A schematic representation of the sonar view (left), and a sonar image output from DIDSON (right) (from Fig. 2 in Han and Uye (2009)).

2.4. Monitoring methods for sessile/sedentary and mobile fauna

2.4.1. Visual transects

Visual transects are one of the most commonly used methods for monitoring reef fish (Sale and Douglas, 1981; Halford and Thompson, 1994; Samoilys and Carlos, 2000), and are also used extensively for assessing benthic communities (Lessios, 1996; Beenaerts and Berghe, 2005), providing quantitative data at a medium spatial scale (Sanderson and Solonsky, 1986; Beenaerts and Berghe, 2005). The method involves laying tape/rope down over a designated habitat, for a specified length with multiple replicates, which is then swum by an observer (Hill and Wilkinson, 2004). Variations of the method include the point intercept method, where organisms are only noted at specific points along the transect line, and the line intercept method, where organisms are only noted when they cross the transect line (Hill and Wilkinson, 2004). Enumeration of mobile organisms occurs via direct counting, while sessile organisms are usually estimated by percentage cover (Samoilys and Carlos, 2000; Beenaerts and Berghe, 2005). Visual transects provide a relatively easy method to deploy, however, the time required can be considerable (Table 2.9, Fig. 2.8). Additionally, the accuracy of the method can only be considered moderate. This method would be best suited for recording the abundance of fish and/or mobile pelagic/demersal invertebrates (*i.e.* cephalopods) utilising the artificial reef, and benthic organism on the reefs and in the surrounding area.

Table 2.9. The advantages and limitations of the visual transect method for monitoring the fauna on artificial reefs.

Advantages	***Limitations***
Deployment	
• Visual transects require only minimal equipment (Hill and Wilkinson, 2004).	• The deployment of transects and their use is limited by diver occupational health and safety. • The ability to lay the transect is limited by the structure of the habitat. • A taxonomic expert is required to conduct the fieldwork.
Accuracy	
• Samoilys and Carlos (2000) and Colvocoresses and Acosta (2007) found visual transects to record greater reef fish abundance estimates than the stationary visual census method.	• Both Leujak and Ormond (2007) and Jokiel et al. (2015) found visual transects to record lower species richness estimates for corals than visual and photo quadrats, as well as video transects. • Both Weinberg (1981) and Leujak and Ormond (2007) found the method to capture a lower coral cover estimate than visual and photo quadrats. However, Jokiel et al. (2015) found visual transects to record higher coral cover than both of the methods.

Advantages	Limitations
	• For reef associated fish Riberio et al. (2004) found visual transects to record a higher species richness than the stationary visual census method, but a lower level than the rapid visual technique. • Biases in the method can be introduced by variations in swimming speed and distance from the substratum between observers (Cheal and Thompson, 1997). • Some authors also consider the presence of an observer to potentially alter fish behaviour (Hill and Wilkinson, 2004). • Willis (2001) noted that visual transects underestimate the presence and abundance of cryptic fish species.
Precision	
• The method has a moderate to high level of precision (Sale and Douglas, 1981; Sanderson and Solonsky, 1986; Hill and Wilkinson, 2004).	
Time	
	• The time required to complete a sample, compared to other methods, varies between studies (Leonard and Clark, 1993; Leujak and Ormond, 2007; Jokiel et al., 2015). However, in general Abdo et al. (2004) considered it to be a relatively time consuming method.
Cost	
• As minimal equipment is needed to conduct visual transects the costs are limited (Leujak and Ormond, 2007; Jokiel et al., 2015).	

2.4.2. Video transects

The video transect method is a modification of the visual transect method, where a video camera is used to record the transect and analysed at a later date (Pelletier et al., 2011). Video transects are used for monitoring similar organisms as visual transects and provide quantitative data on a medium spatial scale (Abdo et al., 2004; Hill and Wilkinson, 2004; Pelletier et al., 2011). Like visual transects, video transects can prove to be a relatively easy method to deploy, however, both the time required and cost of the method can be considerable (Table 2.10, Fig. 2.8). Again, the accuracy of the method can only be considered moderate. This method would be best suited for recording the abundance of fish and/or mobile pelagic/demersal invertebrates (*i.e.* cephalopods) utilising the artificial reef, and benthic organism on the reefs and in the surrounding area.

Table 2.10. The advantages and limitations of the video transect method for monitoring the fauna on artificial reefs.

Advantages	***Limitations***
Deployment	
• As videos are analysed at a later time, taxonomic experts are not required to collect the footage (Pelletier et al., 2011). • The videos provide a permanent record of observations and can be reanalysed at a later date if required (Abdo et al., 2004).	• Divers are limited in their depth and duration of deployment by occupational health and safety.
Accuracy	
• Jokiel et al. (2015) and Leujak and Ormond (2007) found video transects to provide higher coral species richness estimates than visual transects.	• The accuracy of video transects for coral cover is variable with Leujak and Ormond (2007) finding the method to record a higher estimate than visual transects while Jokiel et al. (2015) found the opposite result.

Advantages	Limitations
	• Species identification can be more difficult on video compared to *in situ* observations. This is reflected in studies by Pelletier et al. (2011) and Holmes et al. (2013) who found that video transects recorded lower fish species richness than visual transects. • Langlois et al. (2010) and Watson et al. (2010) found that visual transects recorded a lower species richness of reef fish compared to remotely operated underwater video.
Precision • Video transects are generally thought to have a high level of precision for sessile/sedentary organisms (Hill and Wilkinson, 2004), though Jokiel et al. (2015) found it to have a relatively low level of precision for the same fauna. • The method has a moderate level of precision for mobile organisms, such as fish (Holmes et al., 2006; Langlois et al., 2010).	
Time • Video transects are a relatively quick method to undertake in the field (Abdo et al., 2004; Pelletier et al., 2011).	• Considerable laboratory time is required to analyse the videos and thus the overall time can be equal to, and sometimes greater than, the total time required for visual transects (Pelletier et al., 2011; Holmes et al., 2013; Jokiel et al., 2015).
Cost • The overall cost of the method is relatively low due to the limited amounts of time required to collect each sample, particularly when compared to visual transects (Pelletier et al., 2011).	• The equipment costs can be considerable, depending on the camera system used (Holmes et al., 2006; Jokiel et al., 2015).

2.4.3. Manta tow

The manta tow method is primarily used for the monitoring of organisms over broad spatial scales to provide semi-quantitative data (Hill and Wilkinson, 2004) (Miller et al., 2009). The method was initially developed for the monitoring of Crown of Thorns starfish (Chesher, 1969) and involves towing an observer behind a boat at a constant speed, who holds on via a 'manta board' (Fig. 2.5). The observer takes visual estimates, stopping at regular intervals to note them down (Miller et al., 2009). This method is used to monitor the percentage cover of benthic habitats (Rodgers and Cox, 1999; Kenyon et al., 2006; Zhang et al., 2006; Rajamani and Marsh, 2015) and for counts of larger invertebrates (Miller et al., 2009; Shiell and Knott, 2010) and fish (Richards et al., 2011; Miller et al., 2012). The manta tow method provides moderate to effective estimates of both sessile/sedentary and mobile fauna (Table 2.11 and Fig. 2.5). It has been shown to be an easy method to deploy that generally only requires a relatively low financial investment and timeframe to undertake. This method would be best suited for recording the abundance of fish and/or mobile pelagic/demersal invertebrates (*i.e.* cephalopods) and benthic invertebrates (*i.e.* large crustaceans) utilising the areas surrounding artificial reefs.

Table 2.11. The advantages and limitations of the manta tow method for monitoring the fauna on artificial reefs.

Advantages	***Limitations***
Deployment	
• The use of a boat to tow an observer results in large areas being easily sampled (Kingsford and Battershill, 2000).	• The duration to which observers can be towed is limited by occupational health and safety. • The method is limited to use in relatively shallow, clear water (Kingsford and Battershill, 2000; Hill and Wilkinson, 2004).

Advantages	Limitations
Accuracy	
• Manta tows have been shown to provide accurate data for estimating the relative abundance of Crown of Thorns starfish (Moran and De'ath, 1992). • The method has been found to effectively provide estimates of live coral cover (Miller and Müller, 1999).	• Manta tows are only able to determine relative abundances of sessile and sedentary organisms (Moran and De'ath, 1992; Hill and Wilkinson, 2004). • Several authors have shown that this method may not adequately detect cryptic individuals (Fernandes et al., 1990; Moran and De'ath, 1992; English et al., 1997).
Precision	
• Manta tows can provide high levels of precision for Crown of Thorns starfish and coral surveys (Moran and De'ath, 1992; Miller and Müller, 1999),.	• The method provides low and moderate levels of precision for dead coral categorisation and mobile organisms, respectively (Miller and Müller, 1999; Hill and Wilkinson, 2004; Miller et al., 2012).
Time	
• Manta tow surveys can be completed relatively quickly (Kingsford and Battershill, 2000).	
Cost	
• The method specific equipment is inexpensive (English et al., 1997; Kingsford and Battershill, 2000; Hill and Wilkinson, 2004). • The time-efficiency of the method lowers field-based expenses (English et al., 1997; Kingsford and Battershill, 2000; Hill and Wilkinson, 2004) and has been shown to be a relatively cost-effective monitoring option for Crown of Thorns starfish (Moran and De'ath, 1992).	

Fig. 2.5. An observer being towed over coral reef demonstrating the manta tow technique (from Fig. 2 in Miller et al. (2009)).

2.4.4. Towed video

The towed video method is similar to the manta tow technique, where a video camera, rather than an observer, is dragged behind a boat at a constant speed, along a predetermined transect (Machan and Fedra, 1975; Mallet and Pelletier, 2014). Unsurprisingly, both methods sample over the same spatial scale and result in data of the same resolution and quality being generated (Hill and Wilkinson, 2004; Assis et al., 2007). There are a number of variations of the towed video method, primarily based upon the camera's position in the water column (Table 2.12). As outlined in Table 2.13 and Fig. 2.8, the towed video method provides relatively accurate estimates of species richness and abundance for sessile/sedentary and mobile fauna. The method can be carried out quickly but its expense must be taken into consideration. This method would be best suited for recording the abundance of fish and/or mobile pelagic/demersal invertebrates (*i.e.* cephalopods) and benthic invertebrates (*i.e.* large crustaceans) utilising the areas surrounding the artificial reef.

Table 2.12. Variations on the towed video method, including the camera's position in the water column and the types of flora and fauna monitored.

Type of towed video	Position in water column	Organisms monitored	Studies
Seabed tow	Seafloor	Benthic organisms	Machan and Fedra (1975), Spencer et al. (2005), Rooper (2008)
Mid-water tow	Mid-water	Macro-fauna, algae, seagrass, habitat	Riegl et al. (2001), Assis et al. (2007), Norris et al. (1997), Morrison and Carbines (2006), Grizzle et al. (2008), Carbines and Cole (2009), McIntyre et al. (2015)
Video towed diver	Surface or mid-water	Benthic organisms	Jokiel et al. (2015)

Table 2.13. The advantages and limitations of the towed video method for monitoring the fauna on artificial reefs.

Advantages	*Limitations*
Deployment	
• Towed video can be deployed under harsh conditions that restrict the use of human observers. • The video recordings provide a permanent record. • The method is non-invasive.	• The method requires a relatively flat seafloor and high water clarity (Riegl et al., 2001; Grizzle et al., 2008). • Movement of the boat, due to course correction or weather, can impinge on the ability of the camera to stay at a constant depth (Grizzle et al., 2008).
Accuracy	
• By towing a video camera, rather than an observer, this technique is able to sample species that are scared by human interaction (Assis et al., 2007). • Jokiel et al. (2015) found towed video to be one of the more accurate coral monitoring methods for recording their percentage cover.	• The planar view of the cameras can lead to incorrect estimations of benthic cover (Carbines and Cole, 2009). • The technique is limited in its ability to identify small species (Rooper, 2008; Carbines and Cole, 2009).

Advantages	Limitations
• Towed video was found by Morrison and Carbines (2006) to record greater estuarine fish species richness and abundance than visual transects, remotely operated underwater video, fish traps and hook and line methods.	• Whilst the use of automated gear allows for the sampling of fish species affected by the presence of humans, other species may be scared by the gear leading to biases (Morrison and Carbines, 2006; McIntyre et al., 2015). • Jokiel et al. (2015) noted that lower numbers of coral species were recorded when using towed video compared to visual and photo quadrats, and visual transects. • Morrison and Carbines (2006) found towed video to capture lower species richness and abundance of estuarine fish than trawling.
Precision	
	• The method has a low level of precision for sessile/sedentary organisms (Hill and Wilkinson, 2004) and a moderate level for mobile organisms (Assis et al., 2007).
Time	
• The technique has been shown to be a relatively fast way of collecting data in the field (Riegl et al., 2001; Assis et al., 2007; Jokiel et al., 2015).	
Cost	
	• The expense of the gear can vary significantly depending on the level of sophistication required (Hill and Wilkinson, 2004; Rooper, 2008; Carbines and Cole, 2009; Jokiel et al., 2015), making the cost of the method dependent upon the study being conducted.

2.4.5. Remotely operated underwater video

The first underwater video systems, used in the application of marine biology, date back to 1949 (Barnes, 1952) and they have since been used for the monitoring of a variety of organisms including reef (Dunbrack and Zielinski, 2003; Watson et al., 2010; Chabanet et al., 2012) and deep demersal fish (Priede et al., 1994; Priede and Merrett, 1996), as well as benthic organisms (Tyne et al., 2010). There are a variety of modifications to the general method, which relate to the number of cameras and the presence or absence of bait (Table 2.14). They can be utilised to collect semi-quantitative data on fine to medium spatial scales, with stereo-systems also allowing for the measurement of fish lengths. As shown in Table 2.14 and Fig. 2.8, the remotely operated underwater video method can be deployed easily, at a relatively low cost, though the time investment required can be considerable. It provides relatively effective richness and abundance estimates for sessile/sedentary fauna, and moderate estimates for mobile fauna. This method would be best suited for recording the abundance of fish and/or mobile pelagic/demersal invertebrates (*i.e.* cephalopods) utilising the artificial reef, and benthic organism on the reefs and in the surrounding area.

Table 2.14. Types and specifications of remotely operated underwater video.

Type	Specifications	Studies
Linked	Temporary	Tyne et al. (2010)
	Permanent	Aguzzi et al. (2011), Jan et al. (2007)
Autonomous	Baited (Fig. 2.6)	Priede et al. (1994), Priede and Merrett (1996), Langlois et al. (2010), Lowry et al. (2012), Rizzari et al. (2014), Stobart et al. (2007)
	Unbaited	Dunbrack and Zielinski (2003), Francour et al. (1999); (Chabanet et al., 2012); Pelletier et al. (2012)
	Stereo (two cameras) (Fig. 2.7)	Watson et al. (2005), Watson et al. (2010), Langlois et al. (2015)
	Horizontal camera	Ellis and DeMartini (1995)
	Vertical camera	Willis et al. (2000)
	Lights	Bassett and Montgomery (2011)

Table 2.15. The advantages and limitations of the remotely operated underwater video method for monitoring the fauna on artificial reefs.

Advantages	*Limitations*
Deployment	
• The use of a camera, rather than an observer, is less invasive (Cappo et al., 2003). • Video recordings provide a permanent record (Cappo et al., 2003). • Video systems can operate at much greater depths, and for longer periods of time, than any underwater visual census method (Willis et al., 2000). For example, video systems have been successfully deployed in waters up to 4000 meters deep (Priede et al., 1994). • Video deployment requires less people, particularly skilled workers, compared to other methods (Willis et al., 2000; Cappo et al., 2003).	• Permanent linked systems are restricted in their location as after their deployment they are immovable. (Jan et al., 2007; Aguzzi et al., 2011). • Permanent linked systems can be fouled by organisms leading to obscuring of their cameras. • Cameras can only be deployed in areas with high water clarity (Harvey and Cappo, 2000). • The camera field of view can become obscured by environmental conditions, such as vegetated habitats, *ie*. Kelp forests (Willis et al., 2000). • Cameras are limited in their ability to be deployed in caves and structurally non-homogeneous habitats (Watson et al., 2005).

Advantages	Limitations
Accuracy	
• The use of a camera removes biases present in underwater visual census methods due to the potential attraction or repulsion of fish to divers (Willis et al., 2000; Watson et al., 2005). • Baited cameras have been noted as being particularly useful for measuring species richness and relative density of fish (Willis et al., 2000; Watson et al., 2005; Langlois et al., 2010) as they increase the abundance of carnivorous species without negatively affecting the abundance of herbivorous species, compared to unbaited cameras (Harvey et al., 2007). Though this point has been challenged (Watson et al., 2010). • Underwater video has been noted to provide higher reef fish species richness estimates than video transects by both Langlois et al. (2010) and Watson et al. (2010).	• It can be difficult to discriminate taxonomic detail on video, due to the reduction of light with depth, distance of organisms from the camera, and water quality, making species identification difficult (Kingsford and Battershill, 2000; Widder, 2004). • Unbaited video requires a larger number of replicates than baited video methods, due to the chance nature of encountering the target organisms (Watson et al., 2005). • Baited cameras have been noted to underrepresent cryptic species (Watson et al., 2005), site attached fish and those not attracted to bait (Watson et al., 2010). • Baited cameras can only be used to determine relative density because it is incredibly difficult to calculate the area of bait plume dispersal (Priede and Merrett, 1996). • Underwater video has been noted by Rizzari et al. (2014) to capture a lower abundance of reef sharks than the rapid visual technique, but a higher abundance than manta tow, a broad scale method. • Method has been found to record a lower species richness and abundance of fish compared to a range of other methods (Francour et al., 1999; Morrison and Carbines, 2006; Stobart et al., 2007; Frias-Torres and Luo, 2008; Lowry et al., 2012; Rizzari et al., 2014).
Precision	
	• The method has a low level of precision (Watson et al., 2010).

Advantages	Limitations
Time	
• The deployment time of cameras can range from short term, *i.e.* minutes-hours (Tyne et al., 2010) to permanent (Jan et al., 2007; Aguzzi et al., 2011). • Multiple camera drops can be completed in a day (Langlois et al., 2010). • The method requires generally less fieldwork, resulting in lower overall time (Watson et al., 2005).	• The processing time of video footage is considerable (Francour et al., 1999; Harvey and Cappo, 2000; Stobart et al., 2007), though Watson (2006) has shown it is possible to sample the majority of fish species recorded without analysing an entire one-hour drop.
Cost	
• The low amount of time required in the field, to deploy and retrieve the camera equipment, lowers costs. • The method has been cited as being cost effective compared to underwater visual census methods (Cappo et al., 2003).	• The expense of camera equipment can be considerable (Unpublished data by Watson as cited in (Watson et al., 2005)), though the recent application of GoPro cameras allows for a large reduction in costs (Letessier et al., 2015).

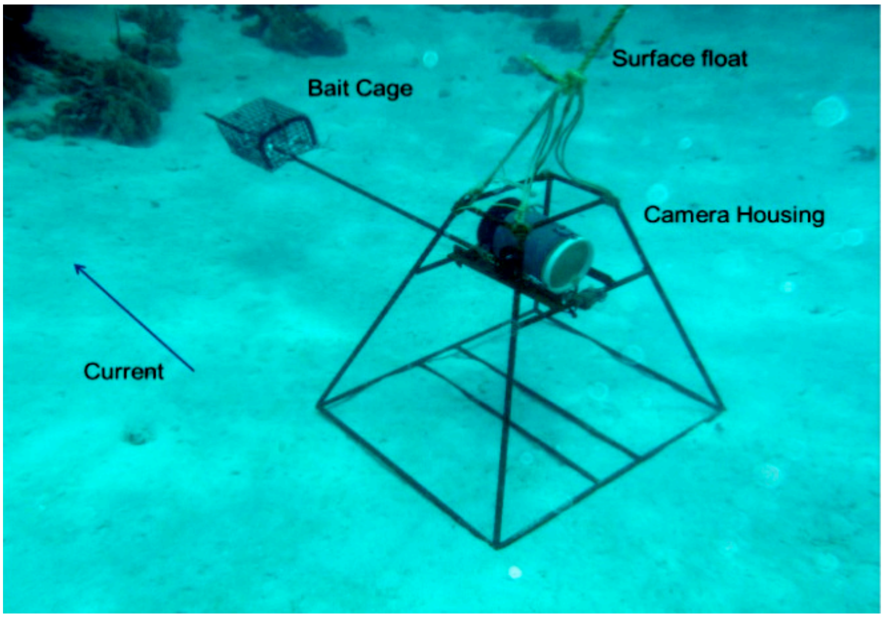


Fig. 2.6. A BRUV system (from Fig. 2 in The Gilis (2015)).

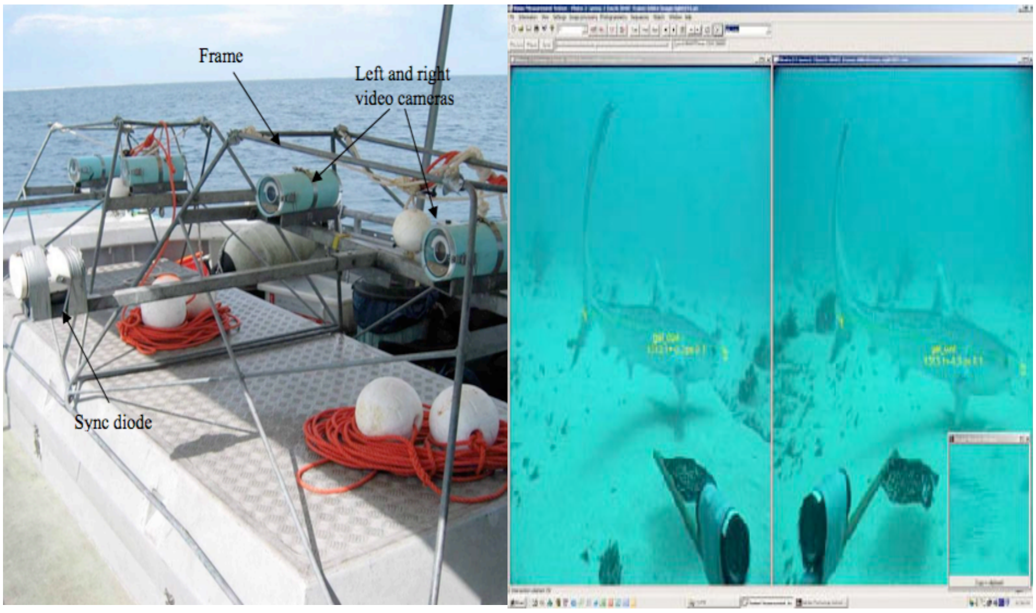


Fig. 2.7. A stereo-video system showing the two cameras on a module (left). Fig. 7 from Watson (2015), and a typical output from such a system allowing for length measurement of fish (right) (from Fig. 1 from America Pink (2016)).

2.4.6. Environmental DNA analysis

Environmental DNA (eDNA) analysis involves the collection and study of genetic material from environmental samples, *e.g.* water (Thomsen and Willerslev, 2015). The method was first developed to analyse the DNA of microbes from sediment (Ogram, 1987), but has since been applied to analysing more complex organisms from a variety of media (Ficetola et al., 2008). Recently the field has focused upon using the method to monitor the presence/absence of endangered and/or invasive species (Jerde et al., 2013; Goldberg et al., 2015; Spear et al., 2015), however, it has the potential to be applied across a broad taxonomic base to sample the DNA of entire communities (Goldberg et al., 2015; Thomsen and Willerslev, 2015). The eDNA method allows for monitoring on a medium to large spatial scale and the gathering of qualitative data. See Thomsen and Willerslev (2015) for an in-depth literature review of the method and its application. The eDNA method, as outlined in Table 2.16 and Fig. 2.8, has proven to be very effective for detecting species richness, of both sessile/sedentary and mobile fauna. However, its inability to record the abundance of organisms must be taken into consideration before it is used in any monitoring program. This method would be best suited for sampling the diversity of organisms, which occur in reference databases, found in the wider vicinity of reef modules.

Table 2.16. The advantages and limitations of the eDNA analysis method for monitoring the fauna on artificial reefs.

Advantages	***Limitations***
Deployment	
• The method only requires small environmental samples be taken (Thomsen et al., 2012).	
• The collection of only environmental sample makes the method relatively non-invasive (Thomsen and Willerslev, 2015; Valentini et al., 2016).	
• The technology can be used to sample structurally complex habitats, such as mangrove systems, where other methods can't be deployed (Thomsen et al., 2012).	
Accuracy	
• The method has been noted as being particularly efficient at sampling cryptic and rare species (Ficetola et al., 2008; Olson et al., 2012; Spear et al., 2015).	• The identification of DNA sequences relies upon reference databases, which are limited in geographical and taxonomic coverage (Kvist, 2013).
• The method has been found to be superior in detecting freshwater fish species over netting (Valentini et al., 2016) and to be the same as, or better than, a number of methods, including netting, hook and line, trawling and underwater visual census methods, in detecting fish species in marine environments (Thomsen et al., 2012).	• The method is unable to reliably estimate abundance, though advancements are being made towards this (Takahara et al., 2012; Kelly et al., 2014; Yamamoto et al., 2016). • There is the potential for contamination using this technique, and therefore potential false positive or negative results (Thomsen and Willerslev, 2015).

Advantages	***Limitations***
	• Methodological artefacts can bias detection towards some taxa and away from others.
	• There is the potential for the detection of species away from their source environment when there is moving water, despite DNA's relatively quick degradation (Thomsen et al., 2012).
Precision	
	• Precision is likely high, though it is only applicable to species richness, as the presence of DNA samples in a liquid medium is likely to be evenly spread, unlike the distribution of organisms.
Time	
• The technique requires little sampling time (Olson et al., 2012; Valentini et al., 2016).	
Cost	
• The method has been noted as being cost-efficient (Olson et al., 2012; Sigsgaard et al., 2015; Thomsen and Willerslev, 2015).	• The technique may require the identification of primers (Valentini et al., 2016), which take time and can be costly.

2.4.7. Extractive methods

Extractive methods rely upon the removal of organisms from their natural environment to ascertain a measurement of catch per unit effort (CPUE) (Cappo and Brown, 1996). This measure is primarily used to assess exploitation levels, but can also be used to estimate relative abundance (Richards and Schnute, 1986; Connell et al., 1998). There are a wide variety of extractive methods, which can provide quantitative data on a medium to broad spatial scale (Table 2.17). Extractive methods generally prove to be efficient in terms of time and financial considerations, however their deployment can be difficult and their ability to effectively sample species richness is somewhat limited (Table 2.18, Fig. 2.8). These methods would be best suited for recording the abundance of fish and/or mobile pelagic/demersal invertebrates (*i.e.* cephalopods) utilising the artificial reef, and benthic organism on the reefs and in the surrounding area, though the exact extractive method used would depend upon the local conditions.

Table 2.17. Commonly used extractive monitoring methods and their variations.

Method	Variations	Studies
Fish trap	O-shaped Z-shaped S-shaped Rotational Baited	Newman and Williams (1995), Langlois et al. (2015), Harvey et al. (2012b), Whitelaw et al. (1991), Morrison and Carbines (2006), Baumgartner et al. (2006), Fujii (2015)
Trawls	Mid-water Bottom Active netting	Říha et al. (2012), Sajdlova et al. (2015), Harmelin-Vivien and Francour (1992), Rozas and Minello (1997), Morrison and Carbines (2006), Spencer et al. (2005)
Ichthyocide	Rotenone Cyanide	Ackerman and Bellwood (2000), Prochazka (1998), Smith-Vainz et al. (2006), Willis (2001)
Hook and line	Handline Long line Drop-line	Morrison and Carbines (2006), Connell et al. (1998)

Table 2.18. The advantages and limitations of extractive methods for monitoring the fauna on artificial reefs.

Advantages	***Limitations***
Deployment	
• Extractive methods, such as ichthyocide, fish traps, and hook and line methods can be used to sample a wide variety of habitats and depths (Newman and Williams, 1995; Cappo and Brown, 1996). • Trawls can be used to sample large areas (Sajdlova et al., 2015).	• Extractive methods that rely upon the removal of organisms are invasive and can have a considerable negative impact upon the environment (Alverson et al., 1994; Ackerman and Bellwood, 2000; Smith-Vainz et al., 2006). • The ability to minimise the impact of extractive methods through the mark, release and recapture method is limited as it is not reliable for estimating abundances (Cappo and Brown, 1996). • Extractive methods, aside from trawling, sample relatively small areas.

Advantages	Limitations
	• The equipment required for trawling is cumbersome (English et al., 1997) and cannot be deployed in all habitats. • High densities of fish can lead to gear saturation for hook and line methods and result in an incorrect calculation of abundance. The ability of the fisher can also affect the usefulness of the method for sampling (Cappo and Brown, 1996).
Accuracy • Ichthyocides are useful for sampling the entirety of a fish community and are particularly good for collecting small and cryptic species (Ackerman and Bellwood, 2000; Willis, 2001). • Selectivity in fish traps can be useful in the targeting of specific species, or age classes (Wells et al., 2008).	• Trawls (Piasente et al., 2004; Wells et al., 2008), fish traps (Newman and Williams, 1995; Robichaud et al., 1999; Fujii, 2015) and hook and line methods (Cappo and Brown, 1996) can show selectivity of fish in terms of size and life stage. • Some fish species are known to actively avoid ichthyocide plumes, leading to biased sampling (Ackerman and Bellwood, 2000).
Precision • Ichthyocides show a high level of precision (Ackerman and Bellwood, 2000).	• Hook and line, trawl and traps can show considerable variability in their level of precision (Cappo and Brown, 1996; Rozas and Minello, 1997; Watson, 2015).
Time • These methods allow for quick sampling (Cappo and Brown, 1996; Harvey et al., 2012b).	
Cost • The hook and line method is one of the most cost efficient methods in terms of the gear used (Cappo and Brown, 1996).	

2.4.8. Fisher surveys

Fisher surveys are used to estimate recreational fishing effort and the associated recreational fishing-related mortality for a given area (Keller et al., 2016). The surveys are typically conducted in two ways, through onsite and offsite surveys (Hartill and Edwards, 2015). Onsite surveys involve face-to-face interviews with fishers at, or near, the site of activity (*e.g.* a boat ramp), while offsite surveys comprise phone, internet surveys or self-reporting of individuals taken from a larger population (Hartill and Edwards, 2015). This duel approach allows quantitative information to be collected on fine to broad spatial scales. As outlined in Table 2.19 and Fig. 2.8, fisher surveys are generally used sparingly as they prove ineffective in terms of their deployment, time and cost, whilst also only providing minimal effectiveness in terms of richness and abundance estimates. This method would be best suited for estimating recreational fishing effort, catch rates of target species in sites at, or around, artificial reefs, as well as the general experience of recreational fishers.

Table 2.19. The advantages and limitations of the fisher survey method for monitoring the fauna on artificial reefs.

Advantages	***Limitations***
Deployment	
• The use of human based surveys makes the method environmentally non-invasive.	
Accuracy	
• Onsite surveys can provide information, such as catch rates, that cannot be reliably obtained through other means (Smallwood et al., 2011).	• Onsite sampling can be unrepresentative (Hartill and Edwards, 2015). • The method can introduce potential errors in recall and misidentification of species (Ashford et al., 2010; Hartill and Edwards, 2015), as well as purposeful misinformation due to the reluctance by fishers to divulge information.
Precision	
	• The level of precision is unknown.
Time and Cost	
	• The collection of recreational fishing data is relatively time intensive, making it an expensive method (West et al., 2012; Hartill and Edwards, 2015).

2.5. Findings

To summarise, the following findings were drawn from the 'heat map' (Fig 2.8).

Reef types:

- All methods are appropriate for monitoring shallow-water reefs.
- The monitoring of deep-water reefs is suited to methods that do not require underwater observers, *eg*. underwater visual census.
- Nearly all methods are suitable for monitoring the faunas of artificial seagrass, however, remote underwater video and manta tow do not provide accurate information on sessile/sedentary faunas present in this environment.

Scale:

- Generally, the methods that are very effective for broad scale monitoring are not suitable for fine scale monitoring (*eg*. manta tow, towed video, eDNA analysis, DIDSON acoustic survey), and *vice versa* (*eg*. settlement tiles, visual quadrats, photo quadrats, stationary visual census).

Data:

- All methods provide some form of quantitative data, except for the rapid visual technique and eDNA.

Deployment:

- Underwater census methods that do not require equipment, such as stationary visual census and the rapid visual technique, as well as methods that do not require the deployment of observers, such as remotely operated underwater video, DIDSON acoustic surveys and eDNA analysis, are the methods that are easiest to deploy.

Species richness accuracy:

- Manta tow, towed video, remotely operated underwater video and eDNA analysis provide the best species richness accuracy estimates, for both sedentary/sessile and mobile fauna, as they are able to cover large areas or record species from a wide area.

Abundance accuracy:

- The most accurate methods for sessile/sedentary fauna are settlement tiles and visual quadrats as they are able to enumerate organisms on a fine scale. Whilst for mobile fauna manta tow, towed video and remotely operated underwater video methods are the most accurate as they can cover large areas or attract individuals from the wider environment.

Precision:

- eDNA analysis likely provides the highest level of precision, though it is only applicable to species richness, as the presence of DNA samples in a liquid medium is likely to be evenly spread, unlike the distribution of organisms.

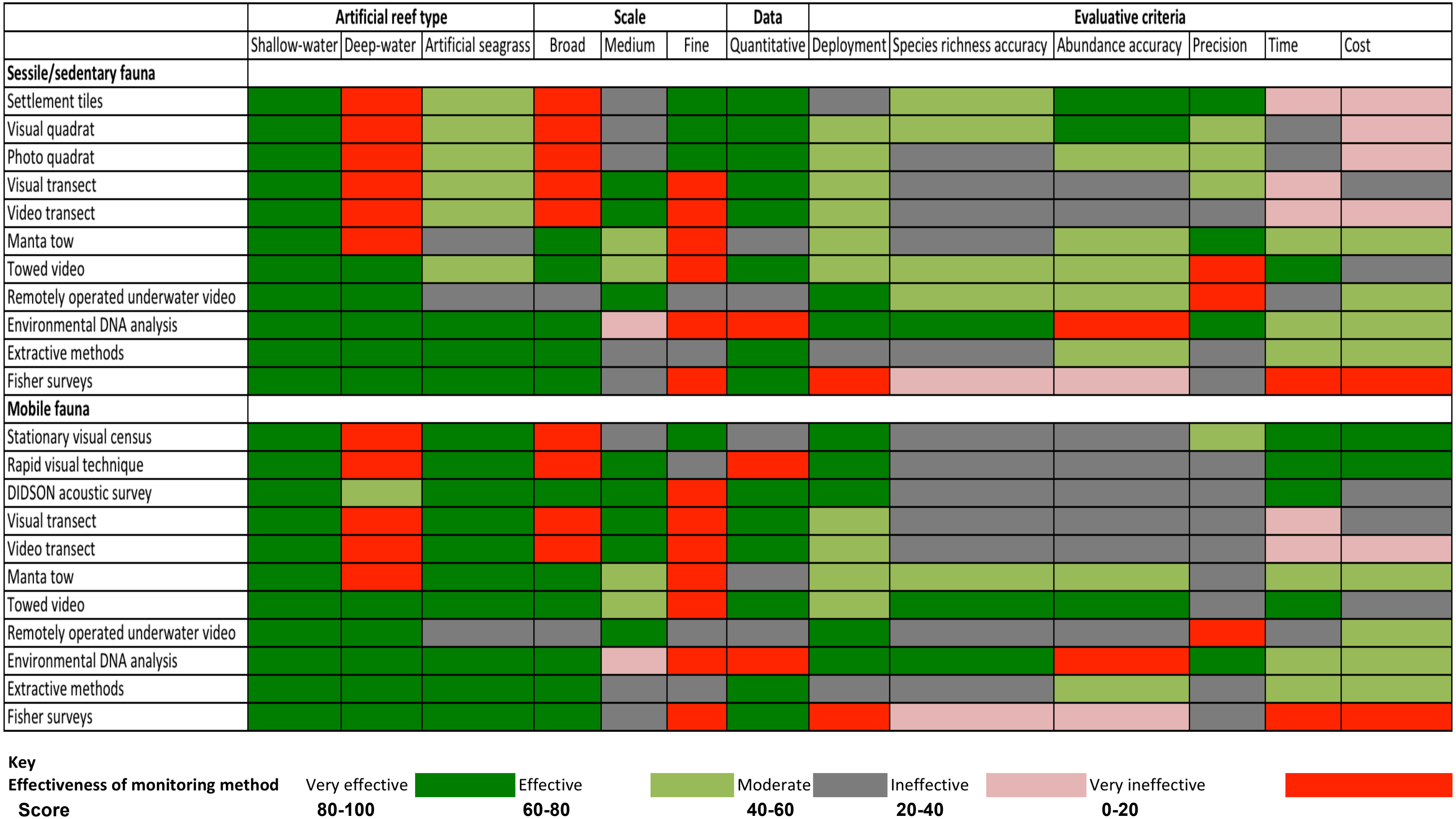
Time and cost:

- With the stationary visual census and the rapid visual technique having both low field and laboratory time, in conjunction with the need for minimal equipment, they have the lowest associated time and cost outputs.

As shown by this review, a plethora of methods can be employed to monitor the fauna (and flora) of artificial reefs. The characteristics of each method are different, meaning that they must be evaluated according to the circumstances of each study, rather than applying a one size fits all approach. As such, this review provides an evaluation of a range of fauna monitoring methods available for artificial reefs.

Of these highlighted methods remotely operated underwater video, which includes BRUV, was found to be easy to deploy, relatively inexpensive and able to provide moderately accurate data on mobile fauna (Fig. 2.8). As such, this technique was utilised as part of a citizen science program in the study of the fish faunas of the artificial reefs of Geographe Bay, which will be covered in Chapter 3.

Fig. 2.8. Heat map showing the relative effectiveness of each monitoring method against different artificial reef types, the scale at which they monitor, the type of data they provide and a range of evaluative criteria. They are ranked on a scale of 0 (red/very ineffective) to 100 (dark green/very effective).



Chapter Three: Characteristics of the ichthyofauna of the Bunbury and Dunsborough artificial reefs determined from Baited Remote Underwater Video

3.1. Abstract

Baited Remote Underwater Video was utilised, through a citizen science based sampling program, to monitor the fish communities of the artificial reefs of Geographe Bay in south-western Australia, between October 2015 and July 2016 inclusive. The resultant videos were analysed to determine if the number of taxa, total MaxN, Simpson's Index, as well as the MaxN of several key recreational species, differed between reefs and over time through a two-way ANOVA, while the composition of the fish communities were tested via a PERMANOVA. Most of the 60 taxa recorded were resident teleosts, while nine species of elasmobranch were also recorded. In terms of the number of individuals, most were either pelagic or epibenthic, and fed on zooplankton or the zoobenthos. Significant differences were found for the majority of variables between reefs, except for the Simpson's Index, with the Dunsborough reef generally displaying higher values. Differences between months were found to be significant for all variables, except for the Simpson's Index and the MaxN of *Chrysophrys auratus*; while the interaction of reef and month were significant for all variables, except for the Simpson's Index and the MaxN of *Chrysophrys auratus* and *Seriola hippos*. Differences between reefs always exerted the greatest level of influence, indicating that the habitat connectivity of the artificial reefs were predominantly responsible for shaping their associated fish communities. The lower, but still significant, differences obtained between months and the reef×month interaction indicated that temperature, as induced by seasonal changes, and oceanographic processes, also played a part in shaping the fish communities of the reefs. These results demonstrate that

BRUVs, deployed through citizen science, can be a useful and cost-effective technique for monitoring the fish faunas of artificial reefs.

3.2. Introduction

The composition of fish communities around the world has been shown to display significant variability between habitats. This has been observed between, and within, structured (*e.g.* reef, algae and seagrass) and non-structured habitats (*e.g.* Bare sand; Molles, 1978; Bell et al., 1987; Heck et al., 1989; Howard, 1989; Sogard and Able, 1991; Gray et al., 1998; Jenkins and Wheatley, 1998; Harman and Kendrick, 2003; Heck Jr et al., 2003). These differences are likely due to the provision and degree of structure within such habitats, as increased habitat complexity creates more shelter for fish and increases feeding opportunities (Bohnsack, 1989). As a result, habitats with high levels of structure, conferred through habitat complexity and vertical relief, have increased levels of fish diversity and abundance, compared to other habitats (Molles, 1978; Howard, 1989; Harman and Kendrick, 2003).

Artificial reefs mimic natural reefs by providing structurally complex habitats for fish to utilise, resulting in their increased localised abundance through either their attraction, or the production of new individuals (Bohnsack and Sutherland, 1985; Baine, 2001; Simon et al., 2011). They have been deployed around the world as a tool for fisheries management, to increase fish production in both industrialised and artisanal fisheries (Gil Chang et al., 2011; Santos et al., 2011). A number of factors can influence the associated fish communities of artificial reefs, including design aspects (*e.g.* rugosity, complexity, vertical relief, surface area), while their deployment location has also been found to play a significant role (Bohnsack and Sutherland, 1985). The importance of the location at which the reef is deployed is likely due to the degree of connectivity between the artificial reef and surrounding natural reefs, influencing the dispersal of individuals (Molles, 1978; Bohnsack, 1979; Gladfelter et al., 1980; Gascon and Miller, 1981; Walsh, 1985; Bombace et al., 1994). Furthermore, artificial reefs, like natural reefs, display seasonal differences in their fish communities (Hastings et al., 1976; Sanders Jr et al., 1985; Stephens Jr et al., 1994; Fujita et al., 1996; Bortone et al., 1997). These differences are likely driven by changing sea surface temperatures and oceanographic processes affecting recruitment, and in turn the fish communities present (Milicich, 1994; Booth and Brosnan, 1995; Pearce and Pattiaratchi, 1999; Taylor et al., 2013).

3.2.1. Rationale

Twenty five artificial reefs were deployed in Australia between 2009 and 2015, making it one of the peak deployment periods over the past 50 years (Bateman, 2015). Two of these reefs were deployed in April 2013 off Bunbury and Dunsborough in Geographe Bay, Western Australia, in a project known as the South West Artificial Reefs Trial, led by the Department of Fisheries (Western Australia), in conjunction with Recfishwest. These reefs were designed to increase the abundance of recreationally important fish species such as Pink Snapper *Chrysophorus auratus*, Trevally *Pseudocaranx* spp. and Samson Fish *Seriola hippos*, ultimately to provide improved recreational fishing opportunities (Government of Western Australia, 2013).

Under the *Fish Resources Management Act 1994* (Western Australia) and the *Environmental Protection and Biodiversity Act 1999* (Commonwealth) the biological monitoring of artificial reef deployments may be required. This contributed, along with other factors, to the undertaking of an in-depth biological monitoring program of the South West Artificial Reefs Trial by the Department of Fisheries (Western Australia; Department of Fisheries, 2012;

Department of the Environment, 2016b). With future deployments of artificial reefs planned for around the State, including some already underway, there is the need for the identification of cost-effective fauna monitoring techniques, as traditional monitoring programs can be expensive (Cigliano et al., 2015; Edgar et al., 2016). Recfishwest has highlighted this need, and through funding from the FRDC (FRDC-project number 2014/005), has instigated research into cost-effective fauna monitoring options for artificial reefs, from which this research project has developed.

3.2.2. Aims

This project utilised the cost effective method developed by Florisson (2015) and Bateman (2015) of deploying BRUV units, through citizen science, to monitor the fish faunas of the two artificial reefs of Geographe Bay. Specifically, this research aims to:

1. Determine whether the characteristics (including the number of taxa, abundance and composition) of the fish fauna differed between the Bunbury and Dunsborough artificial reefs from October 2015 to July 2016 inclusive.

2. Determine whether the characteristics (including the number of taxa, abundance and composition) of the fish fauna differed on a temporal scale between October 2015 and July 2016 inclusive, for both the Bunbury and Dunsborough artificial reefs.

3. Determine what groups of fish utilised the Bunbury and Dunsborough artificial reefs, and how, by assigning the observed taxa to habitat usage, feeding and residency guilds.

Following these aims, it was hypothesised that the fish communities of the artificial reefs of Geographe Bay will display both spatial and temporal differences, due to the influence of habitat connectivity, sea surface temperature and oceanographic processes. The data obtained from this study helped to increase our understanding of how fish utilise these reefs, in what abundances and during which parts of the year, as well as providing a judgement on the efficacy of BRUV technology, deployed through citizen science, for monitoring the fish faunas of artificial reefs.

3.3. Materials and methods

3.3.1. Study site

Geographe Bay is a protected marine embayment located 270 km south of Perth in south-western Australia (Figure 1; Bellchambers et al., 2006). It extends from the Bunbury breakwater in the north to Cape Naturaliste in the south-west, covering an area of approximately 470 km^2 (White et al., 2011). The bay has a relatively shallow bathymetry (maximum depth of 30 m) with a predominantly sandy substratum (Australian Government, 2008). Extensive seagrass meadows, primarily composed of *Posidonia sinuosa*, cover 60% of the bay and are estimated to provide over 80% of the benthic primary productivity for the area (McMahon et al., 1997; Australian Government, 2008). However, eutrophication from catchment runoff has led to accelerated algal growth, resulting in reduced seagrass cover in recent years (Australian Government, 2006).

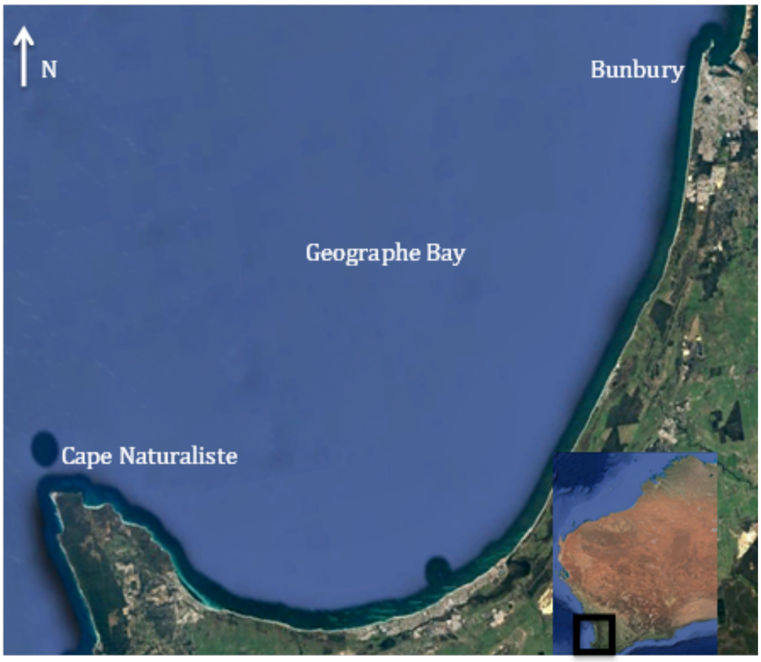


Fig. 3.1. Satellite image of Geographe Bay, denoting its boundaries as marked by Bunbury in the north and Cape Naturaliste in the south-west. Inset image shows the location of Geographe Bay in Western Australia (Google, 2016).

South-western Australia experiences a Mediterranean climate, characterised by hot, dry summers and cool, wet winters (Hodgkin and Hesp, 1998). Rainfall is highly seasonal, with 60-70% falling between May and September, although the amount of rainfall has decreased markedly in the past 40 years (Hodgkin and Hesp, 1998; Timbal et al., 2006). The water temperature in the bay varies between a minimum of 14.8°C in winter to a maximum of 21.6°C in summer (McMahon et al., 1997).

Due to a lack of large rivers and estuaries feeding into the bay, its salinity remains close to full strength seawater throughout the year (Gallop et al., 2012). This minimal fluvial input, in conjunction with the low magnitude diurnal tide regime found in the region, and the presence of an extensive chain of offshore limestone reefs, results in the hydrodynamics of the local water column being driven predominantly by local wind processes (Pattiaratchi et al., 1997). Of these winds, the south-westerly summer sea breezes are the strongest (Pattiaratchi et al., 1997; Smale, 2012). The presence of these winds in the region would be expected to produce upwelling, however the Leeuwin Current suppresses this process (Twomey et al., 2007; Waite et al., 2007). This contributes to the oligotrophic nature of the region, resulting in reduced primary productivity compared to areas which experience upwelling (Twomey et al., 2007). The strength of the Leeuwin Current, which promotes the poleward movement of low nutrient warm water along the continental shelf, varies seasonally, with strongest flow during autumn and winter, and weakest flow during summer (Godfrey and Ridgway, 1985; Twomey et al., 2007; Waite et al., 2007). The weakening of the Leeuwin Current during summer facilitates the equatorward flow of the cold Capes Current and promotes the flushing of Geographe Bay (Pearce and Pattiaratchi, 1999; CoastWise, 2001).

A recent BRUV-based survey of the fish communities of Geographe Bay recorded 76 fish species, of which the most abundant were Western Striped Grunter *Pelates octolineatus*, Yellowtail Scad *Trachurus novaezelandiae* and Skipjack Trevally *Pseudocaranx wrighti* (Westera et al., 2007*)*. Differences in fish communities were noted on small spatial scales throughout the bay as well as with distance from the shore. For a number of reef associated species, including Western King Wrasse *Coris auricularis*, higher abundances were noted at sampling sites in close proximity to rocky reefs, such as near Cape Naturaliste, while their abundance fell at sites further away. Additionally, the bay supports a number of commercial fisheries, including beach seine and gill net fisheries targeting Yelloweye Mullet *Aldrichetta forsteri*, Australian Herring *Arripis georgianus,* Western Australian Salmon *Arripis truttaceus*, Sandy Sprat *Hyperlophus vittatus*, Flathead Grey Mullet *Mugil cephalus*, Western Sand Whiting *Sillago schomburgkii* and the Fringe-Scale Round Herring *Spratelloides robustus*, as well as a trawl scallop fishery (Government of Western Australia, 2012). Boat-based recreational fishing effort for the area has been designated as low to moderate (Sumner and Williamson, 1999).

3.3.2. Geographe Bay artificial reefs

The Geographe Bay artificial reefs, which form the South West Artificial Reefs Trial, were deployed off Bunbury and Dunsborough in April 2013 by the Department of Fisheries (Western Australia). Each reef is comprised of 30 'Fish Box' modules, placed in six clusters of five modules, deployed over a four-hectare area (Fig. 3.2). Each module, which measure 3 m^3 and weighs 10 tonnes, is constructed from steel-reinforced concrete with curved cross braces designed to promote upwelling (Haejoo, 2016). The centre point of the Bunbury artificial reef is located at 115° 35.900'E 33° 18.500'S and lies in ~17 m of water, while the Dunsborough artificial reef centre point is located at 115° 9.980'E 33° 33.962' S at ~27 m depth (Fig. 3.3). Both reefs are deployed within 5 km of boat ramps to allow for easy boat based access by recreational fishers (Florisson, 2015).

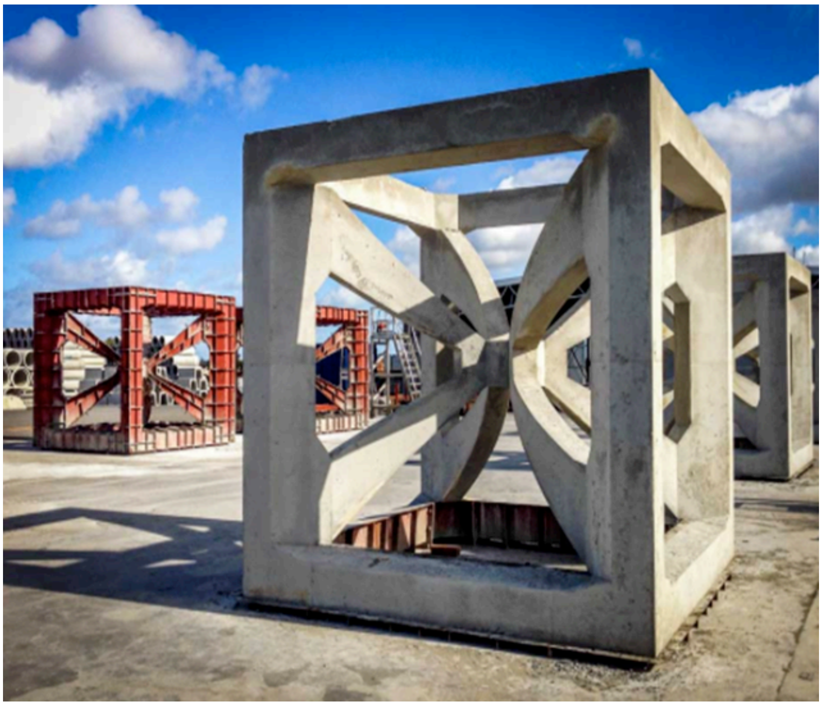


Fig. 3.2. An image of a 'Fish Box' unit, deployed as part of the South West Artificial Reefs Trial (Haejoo, 2016).

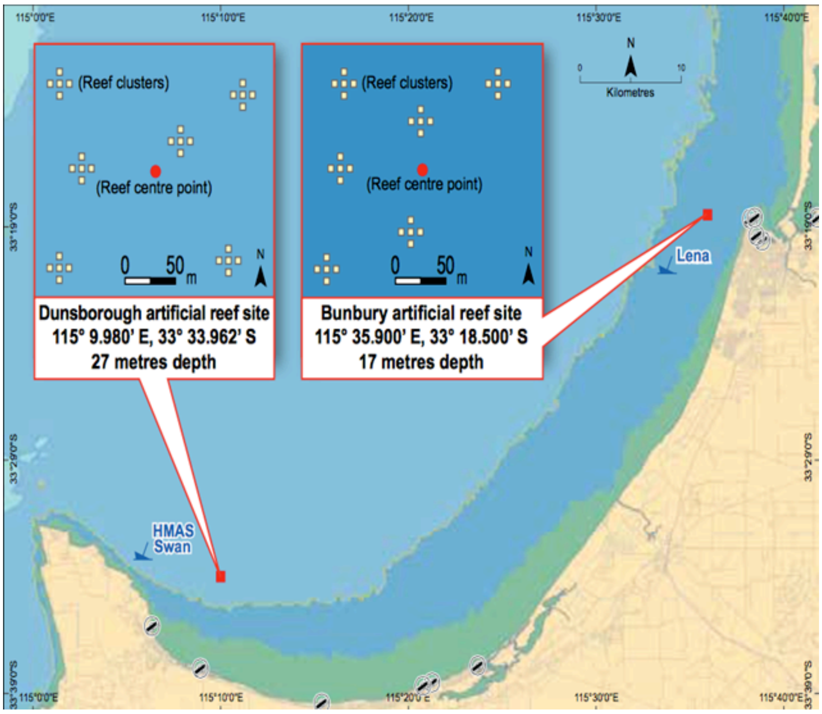


Fig. 3.3. A map displaying the location and co-ordinates of both the Bunbury and Dunsborough artificial reefs (Government of Western Australia, 2013).

3.3.3. Sampling regime

This project utilised underwater video footage, via BRUV, obtained through the 'Reef Vision' citizen science program, run by Recfishwest and Murdoch University. In this program recreational fishers, who lived in close proximity to the reefs and fished them regularly, were recruited via a targeted media campaign. Each potential participant was interviewed to ensure their suitability and, if selected, attended a short training workshop. These measures were undertaken to increase volunteer engagement and reduce attrition rates from the program (see Florisson, 2015). At the workshop, participants were provided with a BRUV unit, data storage devices, prepaid envelopes and bait vouchers. As the study by Florisson (2015) indicated, communication between the volunteers and the project managers/scientists was vital to maintaining their engagement. To encourage this a closed Facebook page was created and all participants invited to join (Fig. 3.4). This provided a platform for volunteers to interact with each other and the project staff by sharing photos and videos of their experiences, as well as discussing various topics and research findings (Tweedley et al., 2016).

A total of 12 primary volunteers were originally recruited, split evenly over both reefs. Two volunteers withdrew from the program part way through the year, whilst an additional volunteer was recruited, resulting in 11 volunteers overall, five for the Dunsborough reef and six for the Bunbury reef. However, videos were only received from nine participants. Sampling commenced in October 2015 and ran for one year, up to and including September 2016. Volunteers were asked to collect two, one-hour long video drops per month on their allocated reef, with the intention of collecting at minimum three videos per reef, per month. In an effort to maintain volunteer engagement, participants were allowed to choose the timing, within each month, and location, within their allocated reef, for the deployment of the BRUV units.

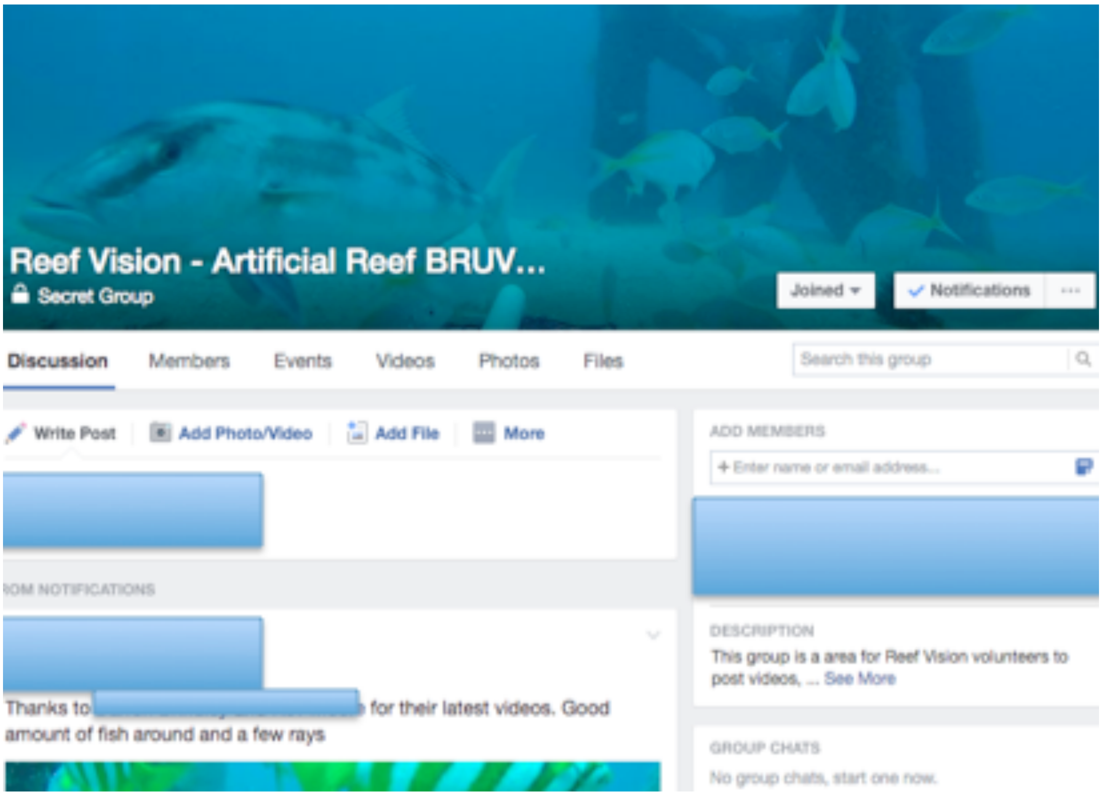


Fig. 3.4. A print screen of the 'Reef Vision' Facebook page, which facilitated communication between volunteers and project staff. Note that all volunteers names have been removed as per Human Ethics requirements (Facebook, 2016).

The BRUVs provided to the volunteers were designed, and purpose-built, for the study by Ecotone Consulting. Each unit was comprised of a 'sled' like frame, a plastic mesh bait bag and a GoPro Hero 4 Silver Action Video Camera™ placed inside a waterproof housing with a Battery BacPac ™ (Fig. 3.5). The GoPro™ camera provided 1080 pixel resolution at 60 frames per second (GoPro Inc, 2016). The frame was constructed from Polyvinyl chloride pipe glued together using plumbers cement. The skids of the frame contained four 680 g lead weights, which provided stabilisation for the unit to ensure that it was deployed in an upright position. The bait arm was suspended 150 millimetres (mm) above the substratum and 600 mm from the centre of the BRUV unit. The mesh bait bag, which was held on the arm, measured 180×100 mm and was placed 500 mm in front of the camera. Volunteers were instructed to place one kilogram of Australian Sardines (*Sardinops sagax*) into the bait bag before deployment. This type of bait is routinely used in BRUV studies (Watson et al., 2005; Harvey et al., 2007; Stobart et al., 2007; Langlois et al., 2010; Watson et al., 2010; Langlois et al., 2015) as its oily flesh attracts a greater abundance of fish, compared to white flesh bait (Dorman et al., 2012).

Each volunteer was provided with a waterproof logbook and asked to record their name, the date on which the sampling took place, the reef location (*i.e.* Bunbury or Dunsborough), as well as the co-ordinates and/or the cluster on which the drop occurred (see Fig. 3.3), the time the BRUV unit was deployed and retrieved, number of boats in the area and any additional comments. Following deployment, the video footage was downloaded from the camera by the volunteer onto a USB stick and posted to Murdoch University for analysis. The USB stick was then returned via mail together with a bait voucher, prepaid envelope and a personal message of thanks, as well as a comment on the contents of the video.

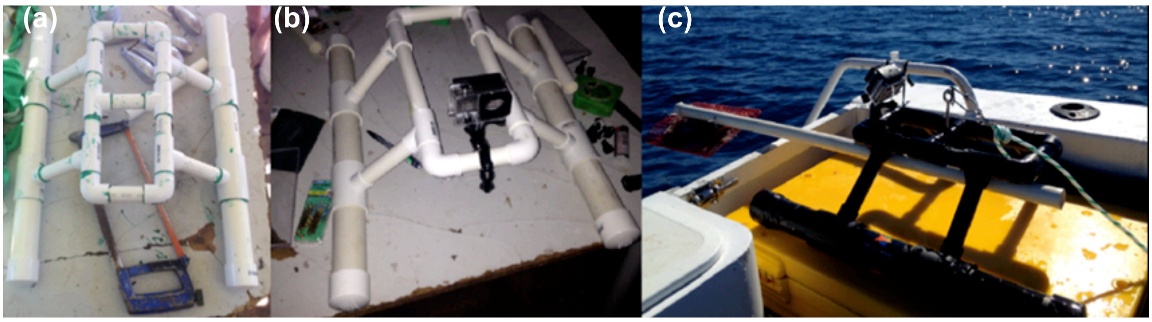


Fig. 3.5. A series of photographs showing the components and construction of the BRUV; (a) gluing of the sled, (b) mounting of the camera and (c) the completed BRUV prior to deployment (Florisson, 2015).

3.3.4. Data collection

Prior to analysis, each video was examined to determine the quality of the footage. Videos in which the alignment of the camera was altered (*i.e.* facing into the sediment or towards the surface of the water) were excluded, as were those in which the water and/or light clarity precluded the identification of fish. Following this a random subsample of four videos from each reef, in each month, were selected for analysis. However, there were two exceptions. The first was for the Bunbury reef in October 2015, when only three videos were collected and subsequently analysed (Table 3.1). This was a result of data collection only beginning part way through the month. The second was in June 2016 when poor weather caused difficulties in collecting data (Table 3.1). As a result, the data from this month, for both reefs, was combined with that collected in July to create a pooled June/July sample, following which four videos, per reef, were randomly selected and analysed. It is noted that it often took several weeks for video footage collected by the volunteers to reach the researchers, and therefore only data from October 2015 to June/July 2016 was analysed due to time constraints.

Each selected video was analysed, with the number of individuals of each fish and cephalopod species being recorded. Cephalopods were included in this study as they have been identified as one of the primary targets of recreational fishers in Western Australia (Government of Western Australia, 2015). The analysis of each video involved counting the MaxN, the maximum number of individuals from each species recorded in the field of view of the camera at any one time (Cappo et al., 2003). This abundance measure is employed ubiquitously in BRUV studies, as it provides an index of relative abundance whilst eliminating the chance of double counting (Babcock, 2000; Cappo et al., 2003). The MaxN of each species was recorded in five-minute intervals from when the BRUV touched the substrate, initially for 60 minutes as part of the preliminary analysis, and then for 45 minutes for the rest of the study (see Chapter 3.4.1). Taxa were identified to the lowest possible taxonomic level, usually to the species level, with staff from Recfishwest providing advice on identification when required. Fish that could not be identified to the family level, either due to their distance from the camera or from obscuring by physical (*e.g.* seagrass and the reef modules) or environmental conditions (*e.g.* turbidity and water clarity), were excluded from the study.

3.3.5. Allocation of species to guilds

Each taxon recorded during the study was assigned to a number of guilds, namely residency, habitat, and feeding guilds, as well as being given a recreational fishing status. Identified taxa were assigned to residency guilds depending upon how frequently they were observed in the video samples. Assigning mobile taxa to residency guilds is common in artificial reefs studies; however, there is not a single widely accepted categorisation method (*e.g.* Relini et al., 1994; Stephens Jr et al., 1994; Santos et al., 2005; Gül et al., 2011). Thus, to allow comparison with other studies a number of different definitions were combined to create a tiered system. The following residency guilds were used; **Transient:** Taxa that are never present in more than two consecutive months (Costello and Myers, 1996), **Resident level three**: Taxa which occurred in two or more consecutive months (Talbot et al., 1978), **Resident level two**: Taxa present in 50% or more of monthly samples (Relini et al., 1994), and **Resident level one**: Taxa present in at least 87.5% of monthly samples (Costello and Myers, 1996). Note that observations from both reefs were combined to create monthly averages from which the level of residency was assigned.

Each taxon was also assigned to a habitat guild based upon the area within the water column that they primarily inhabit, their position relative to the artificial reef modules and their behaviour recorded on the video footage. The habitat guilds used were taken from Nakamura (1985), which have been adopted as a benchmark for studies of fish assemblages of artificial reefs (see Bombace et al., 1994; Relini et al., 2002), who provided criteria for these guilds, while Wartenberg and Booth (2015) named them. The following guilds were used; **Benthic**: Taxa that primarily had contact with the reef surface, or occupied the reef structure, **Epibenthic:** Taxa that associated with the reef, but rarely made direct contact, and **Pelagic**: Taxa that tended to swim above the reef in the middle, and upper parts, of the water column. Note, traditionally benthic organisms, such as rays, were included in the benthic guild though their limited contact with the reef structure would have placed them in the epibenthic guild.

Additionally, feeding guilds were assigned to each species on the basis of the food resources they utilise, as determined from FishBase and the scientific literature (Froese and Pauly, 2016). For species whose diet and feeding behaviour was unknown, their categorisation was based upon closely related species. The feeding guild definitions in this study were taken from Elliott et al. (2007) who applied them to categorise seven guilds of estuarine fish species. These were; **Detritivore**: Taxa that feed on decaying organic matter and associated organisms, **Herbivore**: Taxa that consume plant material, including those that feed on phytoplankton, **Omnivore**: Taxa that feed on both plant and animal material, **Zooplanktivore**: Taxa that feed primarily on small crustaceans in the water column, **Zoobenthivore**: Taxa that feed on animals that live in, on, or immediately above the substratum, **Piscivore**: Taxa that feed predominantly on fish, and **Opportunist**: Taxa whose feeding behaviour and food preferences will likely change depending on food availability, thus consuming a wide variety of prey.

Finally, species were designated as **targeted** or **non-targeted**, based upon the extent to which they are targeted by recreational fishers. This subjective classification was provided by expert opinion from staff at Recfishwest, the peak body for recreational fishing in Western Australia.

3.3.6. Statistical analysis

3.3.6.1. Preliminary analysis

A pilot study was conducted to ascertain the length of each video that needed to be analysed to provide a robust assessment of the characteristics of the fish fauna present at the time of sampling. The purpose of this analysis was two-fold, firstly, to determine whether the BRUV deployment of one hour was sufficient and, if so, whether the data extraction (*i.e.* calculating MaxN for each species present in a video) could be done over a reduced duration of time, *i.e*. less than the total deployment time. This was done to potentially reduce the amount time required for analysis, as the number of videos received and their total duration was significant (*i.e*. 111 videos totalling ~10,000 minutes of footage; Table 3.1).

Table 3.1. The total number of BRUV videos received from the volunteers for both the Bunbury (Bun) and Dunsborough (Dun) reefs in each month and the total number of minutes of video recorded.

	Oct	Nov	Dec	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Tot.	Min.
Bun	3	14	5	8	7	6	4	5	1	5	1	5	59	5611
Dun	5	4	7	8	6	6	4	5	2	5	4	1	52	4243
Tot.	8	18	12	16	13	12	8	10	3	10	5	6	111	9854

The pilot study, which was undertaken at the start of the project, was based on a suite of 10 videos, with each one being randomly selected from each reef, in each month, between October 2015 and February 2016. The MaxN of all fish and cephalopod taxa in each video was recorded in five-minute intervals, for 60 minutes. This provided an estimate of how the fish fauna recorded changed over time following the deployment of the BRUV and the dispersal of the bait plume.

The MaxN data were subjected to the DIVERSE routine in PRIMER 7 (Clarke and Gorley, 2015) to determine the number of taxa, total MaxN (*i.e.* the sum of the MaxN value for all taxa) and the Simpson's Index for each time interval in each replicate video. The resultant values for each univariate metric were then averaged to provide a single value for each five-minute interval at each reef and plotted as separate rarefaction curves.

Changes in species composition throughout time on each reef were also examined. In this case, the MaxN values of each species in each five-minute interval at each reef were square root transformed to down-weight the contribution of relatively abundant taxa, compared to those with lower MaxN values (Clarke and Green, 1988; Veale et al., 2014). The transformed data were then averaged across replicates for each reef for each five-minute interval and used to construct Bray-Curtis resemblance matrices. The matrix for each reef was firstly subjected to hierarchical agglomerative clustering (CLUSTER; Clarke et al., 2014b) to determine the sites that were 95% similar in terms of their species composition. Each matrix was also used to construct a non-metric Multi-Dimensional Scaling (nMDS) ordination plot (Clarke, 1993), which provides a visual representation of the differences in fish communities with time for both reefs. Circles denoting sites that had a similarity of 95% were then overlaid onto the nMDS plot.

The square-root transformed MaxN data for each time interval on both reefs were used to construct a shade-plot to visualise the trends exhibited by the abundances of the various taxa over time on each reef (Clarke et al., 2014a). The shade plot is a visualization of this averaged data matrix, where a white space for a species demonstrates that the species was not recorded, while the depth and colour of shading, ranging from grey shades through the spectrum to black, represents increasing values for the abundance of that species (Clarke et al., 2014a; Valesini et al., 2014). The averaged samples (on the x axis of the plot) are ordered from lowest to highest time interval for each reef. Fish and cephalopod taxa (on the y axis of the 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., 2014b).

3.3.6.2. Primary analysis

Four videos, for each month, for each reef, were chosen at random and analysed. This involved recording the MaxN of each fish and cephalopod taxon from the moment the BRUV settled onto the substrate until 45 minutes (see Chapter 3.4.1).

3.3.6.2.1 Univariate diversity and abundance indices

The MaxN data for each taxon in each video were subjected to the DIVERSE routine to calculate the number of taxa, total MaxN and Simpson's Index, as well as the MaxN data for three recreational species, *C. auratus*, *Pseudocaranx* spp. and *S. hippos*, to make a single data matrix. Prior to statistical testing each of the above six biotic variables were tested to ascertain if a transformation was required to meet the assumptions of ANalysis Of VAriance (ANOVA), *i.e.* homogeneity of variance and normality. This was achieved by plotting the $\log_e$ mean against the $\log_e$ standard deviation of every group of replicate samples and determining the slope of the relationship, comparing it to the criteria in Clarke et al. (2014b). This analysis indicated that the number of taxa and Simpson's Index required no transformation, the MaxN of *C. auratus* and *Pseudocaranx* spp. needed a square and fourth-root transformation, respectively and for total MaxN and the MaxN of *S. hippos* a log(X+1) transformation was necessary.

Following transformation, each of the six dependent variables was subjected to a two-way ANOVA to determine if the variable differed significantly between reefs (2 levels; Bunbury and Dunsborough) and among months (9 levels; October-June/July) and whether the reef×month interaction was significant. ANOVA tests were conducted using the Statistical Package for the Social Sciences (SPSS; Pallant, 2010). In these, and all subsequent tests, the null hypothesis of no significant difference among *a priori* groups was rejected if the significance level (p) was ≤0.05. When multiple factors in a univariate or multivariate ANOVA (*i.e.* PERMANOVA) were significant, the relative influence of each term in the model was quantified by calculating their contribution to the total of the mean squares. The outputs of the ANOVA tests were then used to select the appropriate factors to display on graphs. Note that untransformed, rather than back transformed, data were used for this purpose as the latter can mask patterns in the data (Rothery, 1988).

3.3.6.2.2. Multivariate analysis of guilds

The proportion of both the number of individuals and species in each of the habitat and feeding guilds were tested to determine if they differed between reefs, among months and whether the reef×month interaction was significant. As these data are multivariate in nature, Permutational Multivariate Analysis of Variance (PERMANOVA) tests were employed, using PRIMER 7 (Anderson, 2001; Anderson et al., 2008). Prior to analysis the percentage contribution data were square-root transformed and used to produce four separate Euclidean distance matrices. Each matrix was, in turn, subjected to two-way PERMANOVA and the results visualised as stacked bar graphs.

3.3.6.2.3. Multivariate analysis of fish community composition

The species composition data (*i.e.* the MaxN of each fish and cephalopod taxon) were transformed with dispersion weighting, followed by a square-root transformation. These transformed data were then used to construct a Bray-Curtis resemblance matrix, which was, in turn, subjected to two-way PERMANOVA using the same design as utilised in the multivariate analysis of guilds.

Dispersion weighting was employed to weight the contributions of highly abundant and rare taxa as many of them differed in their MaxN and the consistency in which they were recorded (Clarke et al., 2014b). This technique specifically downweights taxa whose abundances vary greatly among replicates (*e.g.* schooling species such as *T. novaezelandiae*) compared to those that occur more consistently (*e.g. C. auricularis*). The MaxN scores for each taxon were divided by the mean index of dispersion, *i.e.* the average of the variance to mean ratio in each video, in a particular reef, in a given month. This ensures that while the abundances of taxa differ, each has a similar variability structure (Clarke et al., 2006). A square-root transformation was then employed to balance the contributions of rare and common species.

The dispersion-weighted and square-root transformed data were also averaged over samples for each reef in each month and used to construct a Bray-Curtis resemblance matrix. This was subjected to a nMDS (Clarke, 1993) which provided a visual demonstration of the extent to which fish community composition differed across both reef in each month. The trajectory of the samples between months for each reef were overlain to enable ‘tracking’ of the temporal changes.

A shade plot (see Chapter 3.3.6.1) was constructed from the transformed and averaged data matrix to illustrate the trends exhibited by taxa with respect to reef and month. Note that as 60 species were recorded over the duration of the study, many of which only occurred in a few samples, the shade plot was restricted to those 34 species that had a frequency of occurrence of ≥ 5%. Samples on the x axis were arranged in chronological order separately for both reefs, while the taxa (y axis) were arranged in an order to optimise their seriation.

Finally, nMDS plots for each reef were produced from the Bray-Curtis resemblance matrix to illustrate the temporal changes in fish faunal composition of the six most abundant recreationally targeted fish taxa that occurred on each reef. Segmented bubbles of proportional sizes, representing the dispersion-weighted, square-root transformed and averaged MaxN abundances of the taxa were overlaid on the nMDS to illustrate how the abundances of those key species changed over time on each reef.

3.4. Results

3.4.1. Preliminary analysis

Rarefaction curves for the mean number of taxa, total MaxN and the Simpson's Index, for both the Bunbury and Dunsborough reefs, reached an asymptote prior to the 60 min mark (Figs. 3.6 to 3.8). Approximately 95% of all taxa, and the Simpson's Index, were recorded on the Dunsborough reef after 40 mins, and after 45 mins at Bunbury (Table 3.2). Similar trends were also found for total MaxN, with ~95% of the total value being recorded after 35 mins in Dunsborough and 50 mins at Bunbury.

Table 3.2. Mean percentage number of taxa (% Taxa), total MaxN (%MaxN) and Simpson's Index (%Simp) recorded after analysing BRUV footage from the (a) Bunbury and (b) Dunsborough artificial reefs with time. Percentage values approximately at/or above 95% are highlighted in grey.

Time	(a) Bunbury			(b) Dunsborough		
(mins)	%Taxa	%MaxN	%Simp	%Taxa	%MaxN	%Simp
5	45.6	47.1	70.5	36.2	41.3	79.5
10	54.4	53.8	76.7	40.6	80.2	75.8
15	59.7	64.8	82.5	55.1	87.5	80.1
20	66.7	72.4	83.3	62.3	89.6	83.9
25	73.7	74.8	87.9	69.6	91.5	88.6
30	77.2	82.4	89.7	78.3	93.1	90.9
35	87.7	87.1	91.3	87.0	94.8	92.8
40	91.2	90.5	95.9	95.7	97.1	96.7
45	94.7	91.4	96.5	97.1	97.9	97.6
50	94.7	94.8	98.0	98.5	99.0	99.3
55	100.0	98.1	99.5	98.5	99.2	99.3
60	100.0	100.0	100.0	100.0	100.0	100.0

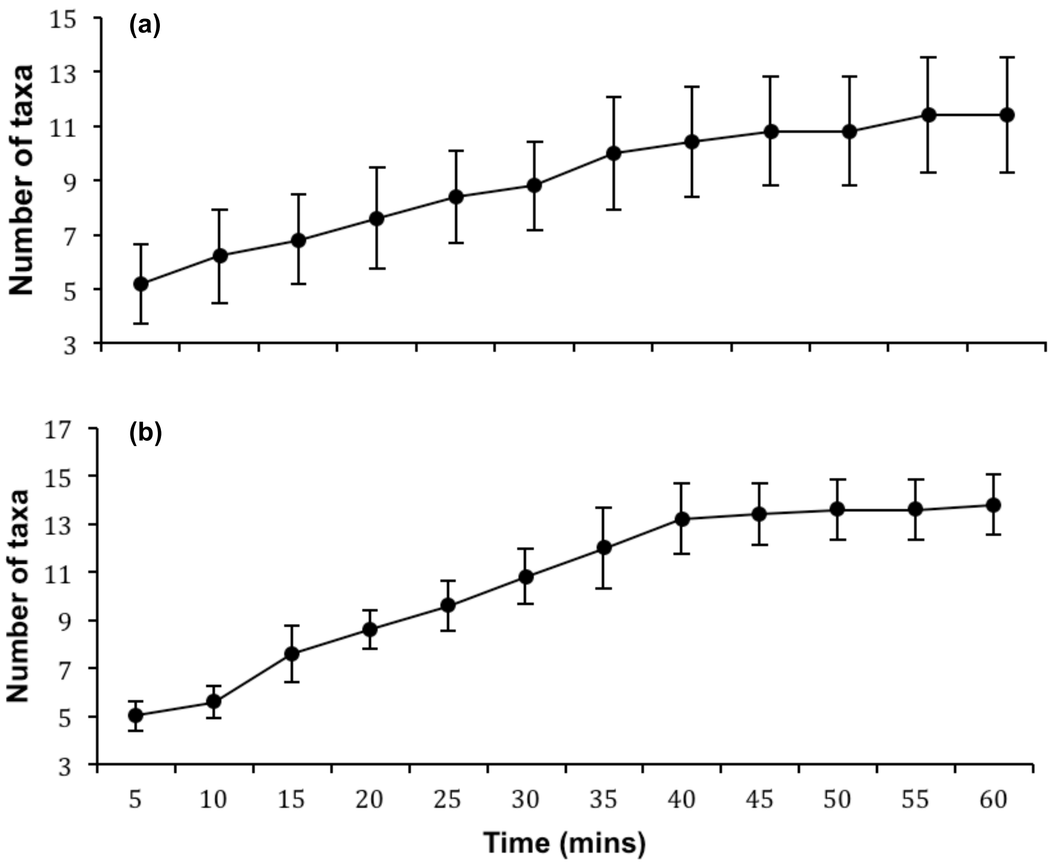


Fig. 3.6. Rarefaction curves for the mean number of taxa *vs* time found on the (a) Bunbury and (b) Dunsborough artificial reefs from BRUV footage. Error bars represent ±1 standard error.

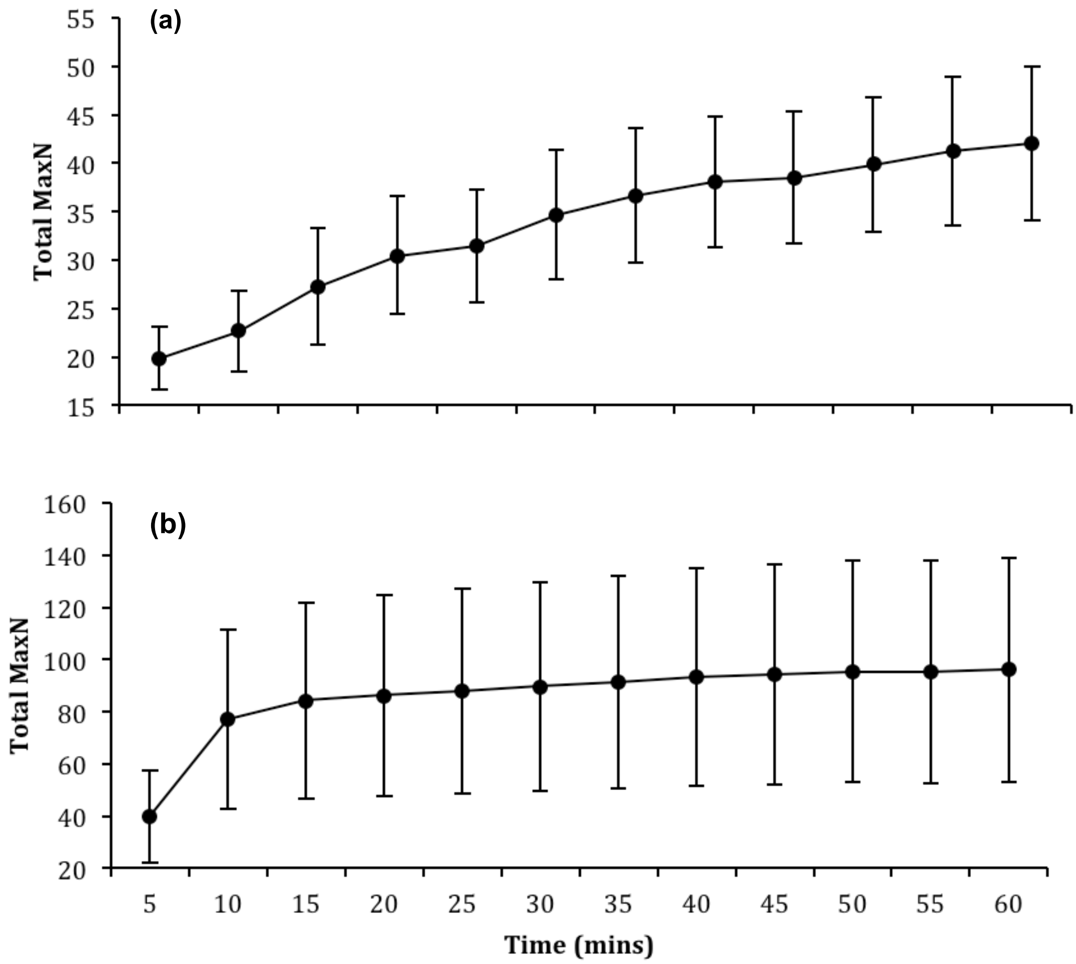


Fig. 3.7. Rarefaction curves for the mean total MaxN vs time found on the (a) Bunbury and (b) Dunsborough artificial reefs from BRUV footage. Error bars represent ±1 standard error.

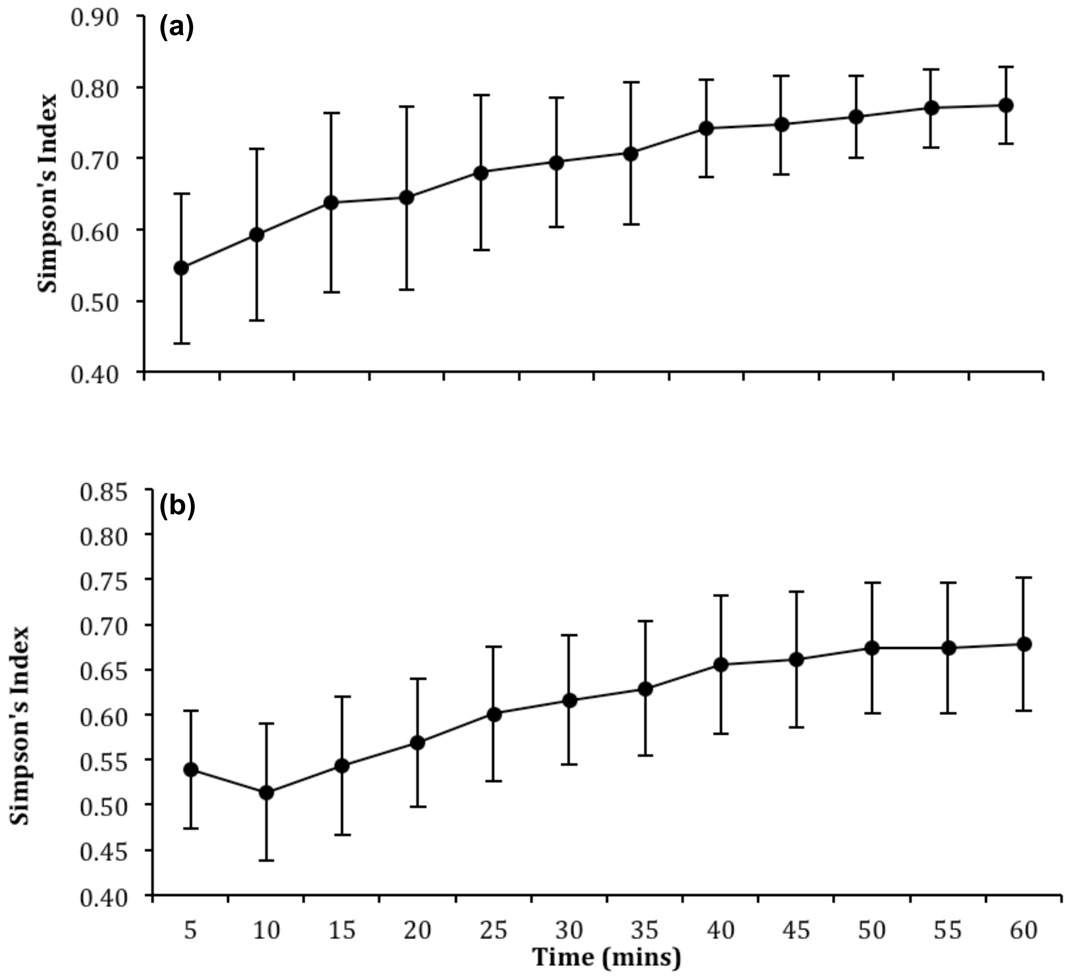


Fig. 3.8. Rarefaction curves for the mean Simpson's Index vs time found on the (a) Bunbury and (b) Dunsborough artificial reefs from BRUV footage. Error bars represent ±1 standard error.

Ordination plots for the square-root transformed fish community data for both reefs showed a sequential progression with time. The lowest amount of time (5 mins) occurred on the left side of the plot, with the largest amount of time (60 mins) on the opposite side (Figs. 3.9a and 3.9b). The distance between pairs of points decreased with time, indicating the faunal composition recorded became more similar as more of the videos were watched and MaxN scored. When circles representing 95% Bray-Curtis similarity were overlain on each of the ordination plots, samples from 35-60 minutes formed a group at the Bunbury reef, and between 40-60 minutes at Dunsborough. Thus, fish fauna composition was 95% similar on both reefs after 40 minutes.

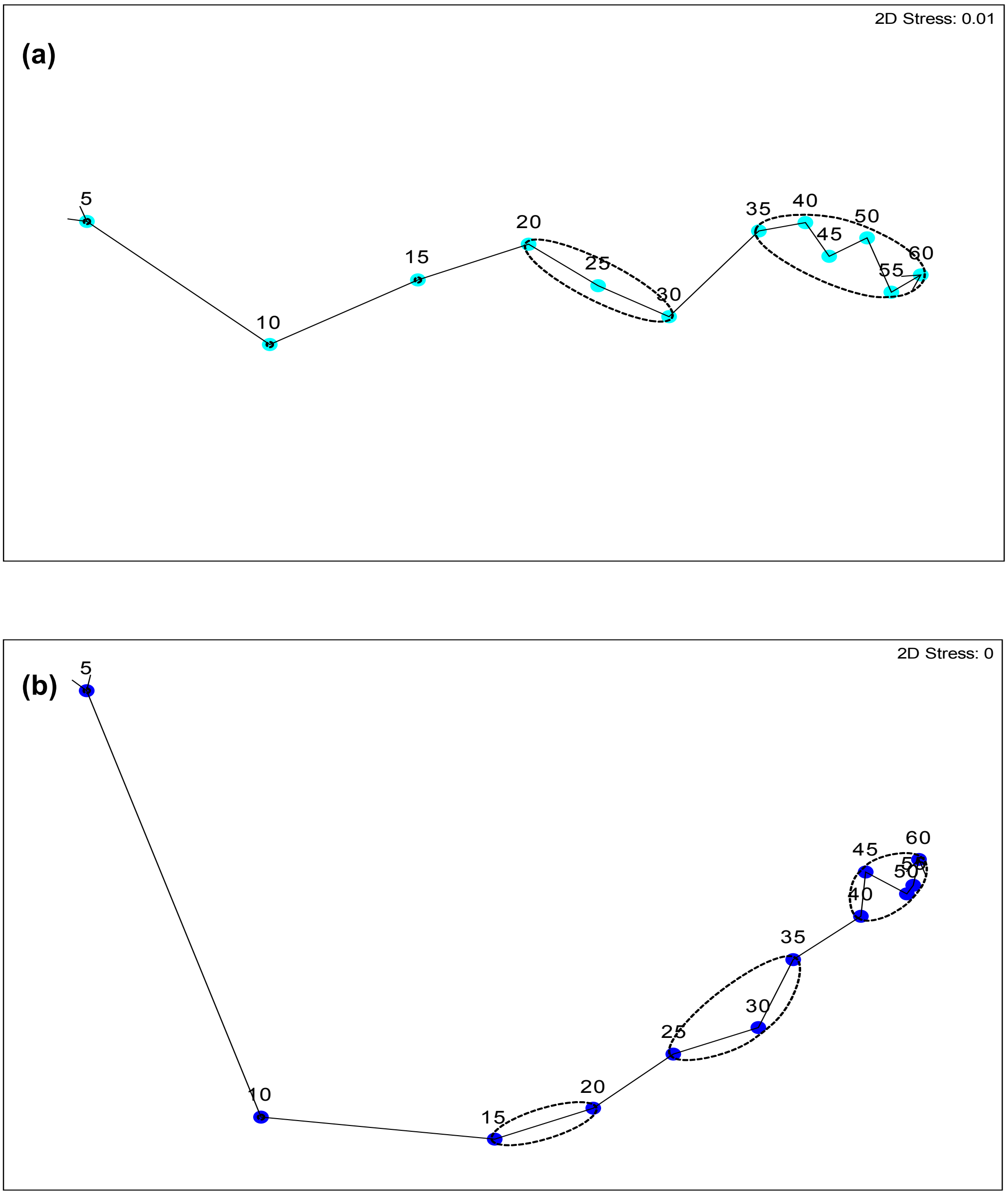


Fig. 3.9. nMDS ordination plots, derived from a Bray-Curtis similarity matrix, constructed from the square-root transformed and averaged MaxN fish community data recorded from consecutive five minute intervals of video footage recorded from the (a) ● Bunbury and (b) ● Dunsborough artificial reefs. Dotted line circles encompass samples that have a Bray-Curtis similarity of ≥ 95%.

The shade plot illustrates that very few new taxa were recorded after 45 minutes and that the MaxN abundance for those taxa that were present typically remained similar (Fig. 3.10). For most taxa, including abundant ones such as the Western Footballer *Neatypus obliquus, C. auricularis* and *T.*

novaezelandiae, their MaxN changed very little over the duration of the video. Moreover, even for those species whose abundance did change with increasing time, *e.g. Pseudocaranx* spp. and Silverbelly *Parequula melbournensis,* these values changed little after 45 minutes on both reefs.

These results demonstrate that it is not necessary to analyse the entirety of a one-hour video to adequately quantify the univariate and multivariate characteristics of the fish fauna of the Bunbury and Dunsborough artificial reefs. As such, only the first 45 minutes of each video will be analysed. This result is supported by the findings of Watson (2006), who conducted a BRUV based study on reef fish communities of the Houtman Abrolhos Islands, Western Australia. She found, from examining the cumulative percentage of species recorded over time, that the majority of species were recorded after 36 minutes of video.

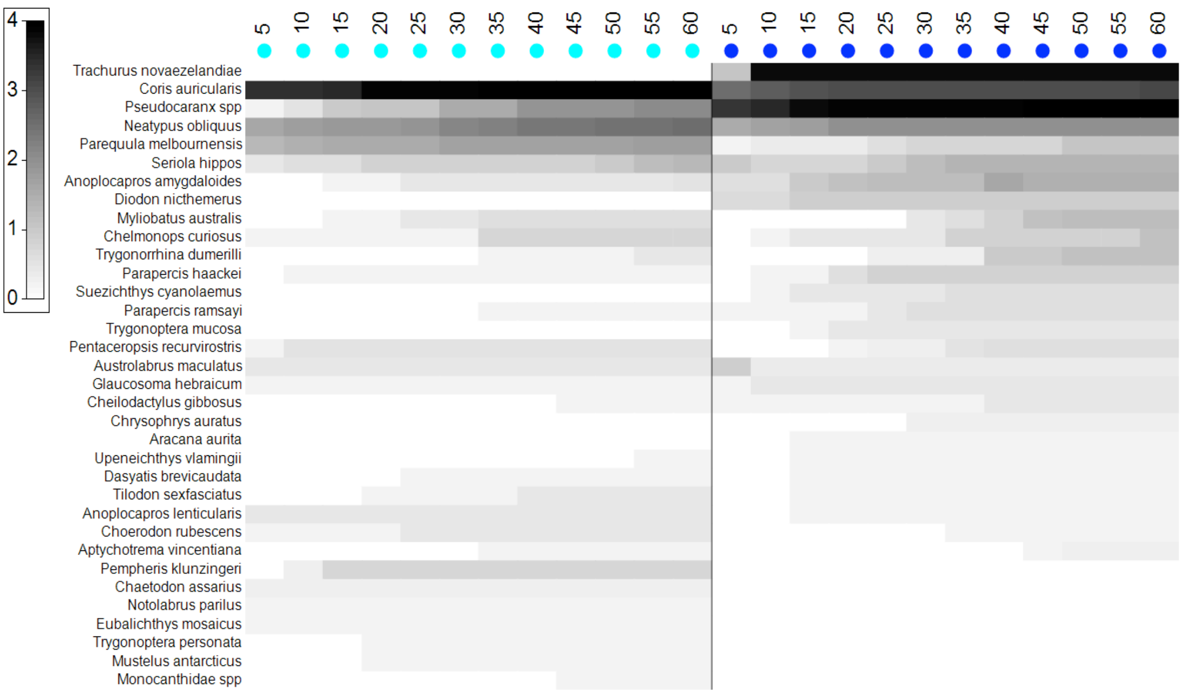


Fig. 3.10. Shade plot, constructed using the square-root transformed and averaged MaxN abundances of each taxa recorded from consecutive five minute intervals of video footage recorded from the recorded in consecutive five minute internals on the ● Bunbury and ● Dunsborough artificial reefs. Numbers above the reef symbols indicate the number of minutes of video that a sample represented. White space denotes the absence of a taxon, with the grey scale represents the transformed and averaged MaxN abundances.

3.4.2. Primary analysis

3.4.2.1. Abundance and frequency of occurrence

Sixty taxa were identified, over the 71 videos analysed, from the Bunbury and Dunsborough artificial reefs, comprising 48 teleosts, nine elasmobranchs and three cephalopods (Table 3.3). Taxa that could not be identified to the species level were either grouped to the family (*i.e.* Pempherididae and Monocanthidae) or genus level (*i.e. Pseudocaranx*). The taxa that contributed the highest proportion to the total MaxN were *N. obliquus* (15%), *Pesudocaranx* spp. (14.2%), *C. auricularis* (14.1%) and the Elongate Bullseye *Parapriacanthus elongates* (12.8%), with all others representing < 5% (Table 3.3). Among the 50 taxa recorded on the Bunbury reef *C. auricularis* was by far the most abundant representing 30.4% of the total MaxN, with *N. obliquus* (11.4%), *P. melbournensis* (10.4%) and *Pseudocaranx* spp. (10%) all occurring in substantial numbers (Table 3.3). A total of 49 species were recorded on the Dunsborough reef, with *P. elongatus* being the most abundant (17.5%), closely followed by *N. obliquus* (16.3%) and *Pseudocaranx* spp. (15.8%).

Overall, the top ten most abundant taxa contributed 81% to the total MaxN, with each one being recorded at the Dunsborough reef and collectively contributing 84% to its total, while nine of the ten (*i.e.* all except, *P. elongates*) were found at Bunbury, representing 71% of its total. Among the recreationally targeted species, *C. auratus* and the Tarwhine *Rhabdosargus sarba* were recorded at Dunsborough and not Bunbury, while the reverse was true for Mulloway, *Argyrosomus japonicus*, although none of these species were particularly abundant (< 0.5% of the mean MaxN).

Among the 60 taxa recorded on either artificial reef, only six, *i.e. C. auricularis*, Western Smooth Box Fish *Anoplocapros amygdaloides*, *S. hippos*, *Pseudocaranx* spp., *P. melbournensis* and the Southern Eagle Ray, *Myliobatus australis* were recorded in more the 50% of the videos (Table 3.4). Of these species, *C. auricularis* was the most frequently recorded appearing in >90% of videos, whereas 26 taxa were recorded in <5% of videos. The top ten taxa in terms of overall frequency of occurrence were also those that were most frequently sighted on each reef, although their values sometimes differed. For example, while *Pseudocaranx* spp. were recorded on 94% of videos at Dunsborough, this taxon only occurred in 49% of videos at Bunbury (Table 3.4). Similar, albeit less marked, trends were found for *N. obliquus* (67% vs 31%) and Western Talma *Chelmonops curiosus* (61% vs 34%). Generally, individual taxa were found less frequently on the Bunbury than Dunsborough artificial reefs, although this was not true for both the Western

Butterflyfish *Chaetodon assarius* (17% vs 0%) and Masked Stingaree *Trygonoptera personata* (11% vs 3%; Table 3.4).

Table 3.3. Habitat (HG) and feeding (FG) guilds, rankings (R), MaxN (N) and percentage contribution to total MaxN (%) for each taxon recorded on the Bunbury and Dunsborough reefs combined and individually on BRUV footage between October 2015 and June/July 2016. The total number of taxa and overall MaxN are also provided. Habitat guild: benthic (B), epibenthic (E) and pelagic (P). Feeding guild: zoobenthivore (ZB), zooplanktivore (ZP), piscivore (PV), omnivore (OV) and herbivore (H). Species targeted by recreational fishers are shaded. * denotes cephalopod taxa

			Overall			Bunbury			Dunsborough		
Taxa	**HG**	**FG**	**R**	**N**	**%**	**R**	**N**	**%**	**R**	**N**	**%**
Neatypus obliquus	E	ZP	1	8.40	15.00	2	3.50	11.40	2	13.20	16.30
Pseudocaranx spp.	E	ZB	2	8.00	14.20	4	3.10	10.00	3	12.80	15.80
Coris auricularis	E	ZP	3	7.90	14.10	1	9.40	30.40	4	6.50	8.10
Parapriacanthus elongates	B	ZP	4	7.20	12.80				1	14.10	17.50
Pempherididae spp.	B	ZP	5	3.10	5.50	8	0.80	2.70	5	5.30	6.60
Trachurus novaezelandiae	P	ZB	6	2.90	5.10	13	0.40	1.20	5	5.30	6.60
Parequula melbournensis	E	ZP	7	2.50	4.40	3	3.20	10.40	12	1.80	2.20
Seriola hippos	P	PV	8	2.10	3.70	5	1.40	4.60	9	2.70	3.30
Pempheris klunzingeri	B	ZP	9	1.90	3.40	7	1.10	3.50	8	2.80	3.40
Diodon nicthemerus	B	ZB	10	1.80	3.20	35	0.10	0.20	7	3.40	4.30
Anoplocapros amygdaloides	E	ZB	11	1.70	3.00	6	1.30	4.30	10	2.00	2.50
Austrolabrus maculatus	E	ZB	12	1.30	2.40	9	0.70	2.20	11	1.90	2.40
Chelmonops curiosus	B	ZB	13	0.70	1.20	10	0.50	1.70	13	0.90	1.10
Trygonorrhina dumerilli	B	ZB	14	0.60	1.10	12	0.40	1.40	14	0.80	1.00
Myliobatus australis	B	ZB	15	0.60	1.10	10	0.50	1.70	16	0.70	0.90
Parapercis haackei	B	ZB	16	0.50	0.90	24	0.10	0.50	14	0.80	1.00
Dasyatis brevicaudata	B	ZB	17	0.40	0.70	20	0.20	0.70	17	0.60	0.70
Cheilodactylus gibbosus	E	OV	18	0.30	0.60	24	0.10	0.50	18	0.50	0.60
Monocanthidae spp.	E	OV	19	0.30	0.60	15	0.30	1.10	24	0.30	0.30
Aracana aurita	E	ZB	20	0.30	0.50	18	0.30	0.80	20	0.30	0.40
Pentaceropsis recurvirostris	B	OV	21	0.30	0.50	20	0.20	0.70	20	0.30	0.40
Notolabrus parilus	E	ZB	22	0.30	0.50	16	0.30	0.90	24	0.30	0.30
Upeneichthys vlamingii	E	ZB	23	0.20	0.40	23	0.20	0.60	20	0.30	0.40
Anoplocapros lenticularis	E	ZB	24	0.20	0.40	20	0.20	0.70	24	0.20	0.30
Parapercis ramsayi	B	ZB	25	0.20	0.40	30	0.10	0.30	20	0.30	0.40
Chrysophorus auratus	E	ZB	26	0.20	0.40				19	0.40	0.50
Argyrosomus japonicus	P	PV	27	0.20	0.30	13	0.40	1.20			
Tilodon sexfasciatus	E	ZP	28	0.20	0.30	30	0.10	0.30	24	0.30	0.30
Chaetodon assarius	E	OV	29	0.10	0.30	16	0.30	0.90			
Choerodon rubescens	E	ZB	30	0.10	0.20	24	0.10	0.50	33	0.10	0.10
Apogon victoriae	B	ZP	31	0.10	0.20	18	0.30	0.80			
Glaucosoma hebraicum	E	ZB	32	0.10	0.20	24	0.10	0.50	33	0.10	0.10
Arripis truttaceus	P	PV	33	0.10	0.20	40	0.00	0.10	28	0.20	0.20
Suezichthys cyanolaemus	E	ZB	34	0.10	0.20				28	0.20	0.20
Aptychotrema vincentiana	B	ZB	35	0.10	0.20	40	<0.10	0.10	28	0.20	0.20
Platycephalus speculator	B	ZB	36	0.10	0.20	30	0.10	0.30	33	0.10	0.10
Chromis westaustralis	E	ZP	37	0.10	0.20				28	0.20	0.20
Trygonoptera personata	B	ZP	38	0.10	0.10	28	0.10	0.40			
Platycephalus longispinis	B	ZB	39	0.10	0.10	30	0.10	0.30	33	0.10	0.10
Rhabdosargus sarba	E	ZB	40	0.10	0.10				28	0.10	0.20
Eubalichthys mosaicus	E	OV	41	0.10	0.10	28	0.10	0.40			
Trygonoptera mucosa	B	ZP	42	0.10	0.10	40	<0.10	0.10	33	0.10	0.10
Eupetrichthys angustipes	B	ZB	43	0.10	0.10	30	0.10	0.30			
Orectolobus maculatus	B	ZB	44	0.00	0.10				33	0.10	0.10
Haliocheres brownfieldi	E	ZB	45	0.00	0.10	40	<0.10	0.10	33	0.10	0.10
Mustelus antarcticus	E	ZB	46	0.00	0.10	40	<0.10	0.10	33	0.10	0.10
Scobinichthys granulatus	E	OV	47	0.00	0.10	35	0.10	0.20			
Octopus tetricus *	E	ZB	48	0.00	0.10	40	<0.10	0.10			
*Sepioteuthis australis**	E	ZB	49	0.00	0.10	35	0.10	0.20			
Achoerodus gouldii	E	ZB	50	0.00	0.10	35	0.10	0.20			
Trygonoptera ovalis	B	ZB	51	0.00	0.10	40	<0.10	0.10			
Sepia apama *	E	ZB	52	0.00	0.10	35	0.10	0.20			
Meuschenia venusta	E	OV	53	0.00	0.10				33	0.10	0.10
Ophthalmolepis lineolatus	E	ZB	54	0.00	0.10	40	<0.10	0.10			
Parupeneus chrysopleuron	E	ZB	55	0.00	<0.10	40	<0.10	0.10			
Enoplosus armatus	E	ZB	56	0.00	<0.10	40			34	<0.10	<0.10
Dactylophora nigricans	E	OV	57	0.00	<0.10	40			34	<0.10	<0.10
Hypoplectrodes nigroruber	E	PV	58	0.00	<0.10	40			34	<0.10	<0.10
Siganus fuscescens	E	H	59	0.00	<0.10	40	<0.10	0.10			
Pseudorhombus jenynsii	B	ZB	60	0.00	<0.10	40	<0.10	0.10			
Total number of taxa				**60**			**50**			**49**	
Total mean MaxN				**56.1**			**30.9**			**80.7**	

Table 3.4. Residency guilds (RG), rankings (R) and percentage occurrence (%) for all identified taxa overall, as well as at the Bunbury and Dunsborough reefs individually, between October 2015 and July 2016 inclusive. Residency guilds: transient (T), resident level three (R3), resident level two (R2) and resident level one (R1). Species targeted by recreational fishers are highlighted. * denotes cephalopod taxa.

		Overall		Bunbury		Dunsborough	
Taxa	RG	R	%	R	%	R	%
Coris auricularis	R1	1	90.1	1	91.4	2	88.9
Anoplocapros amygdaloides	R1	2	78.9	3	74.3	3	83.3
Seriola hippos	R1	3	74.6	2	77.1	4	72.2
Pseudocaranx spp.	R1	4	71.8	5	48.6	1	94.4
Parequula melbournensis	R1	5	64.8	4	60.0	5	69.4
Myliobatus australis	R1	6	52.1	5	48.6	10	55.6
Neatypus obliquus	R1	7	49.3	9	31.4	6	66.7
Chelmonops curiosus	R1	9	47.9	8	34.3	7	61.1
Austrolabrus maculatus	R1	8	47.9	6	37.1	8	58.3
Trygonorrhina dumerilli	R1	10	47.9	6	37.1	8	58.3
Parapercis haackei	R1	11	33.8	19	11.4	10	55.6
Dasyatis brevicaudata	R1	13	28.2	12	20.0	13	36.1
Aracana aurita	R1	12	28.2	10	25.7	14	30.6
Cheilodactylus gibbosus	R1	14	26.8	19	11.4	12	41.7
Notolabrus parilus	R2	15	23.9	10	25.7	21	22.2
Monocanthidae spp.	R2	16	22.5	12	20.0	18	25.0
Pentaceropsis recurvirostris	R2	17	21.1	15	17.1	18	25.0
Upeneichthys vlamingii	R2	18	21.1	15	17.1	18	25.0
Anoplocapros lenticularis	R1	19	21.1	12	20.0	21	22.2
Parapercis ramsayi	R2	20	19.7	24	8.6	16	30.6
Diodon nicthemerus	R2	21	18.3	37	2.9	14	33.3
Tilodon sexfasciatus	R2	22	18.3	24	8.6	17	27.8
Pempheris klunzingeri	R1	23	12.7	19	11.4	25	13.9
Choerodon rubescens	R2	24	12.7	18	14.3	27	11.1
Chrysophorus auratus	R3	26	9.9			23	19.4
Suezichthys cyanolaemus	R3	27	9.9			23	19.4
Glaucosoma hebraicum	R2	25	9.9	19	11.4	28	8.3
Aptychotrema vincentiana	R3	29	8.5	37	2.9	25	13.9
Chaetodon assarius	T	28	8.5	15	17.1		
Trygonoptera personata	R2	30	7.0	19	11.4	36	2.8
Trygonoptera mucosa	R3	31	5.6	37	2.9	28	8.3
Platycephalus speculator	R3	34	5.6	30	5.7	31	5.6
Platycephalus longispinis	R3	32	5.6	24	8.6	36	2.8
Eupetrichthys angustipes	R3	33	5.6	24	8.6	36	2.8
Orectolobus maculatus	R3	39	4.2			28	8.3
Trachurus novaezelandiae	T	35	4.2	37	2.9	31	5.6
Haliocheres brownfieldi	R3	40	4.2	37	2.9	31	5.6
Pempherididae spp	R3	36	4.2	30	5.7	36	2.8
Apogon victoriae	R2	37	4.2	24	8.6		
Eubalichthys mosaicus	R2	38	4.2	24	8.6		
Parapriacanthus elongatus	R3	41	2.8			31	5.6
Chromis westaustralis	R3	43	2.8			31	5.6
Arripis truttaceus	R2	44	2.8	37	2.9	36	2.8
*Octopus tetricus**	R3	46	2.8	37	2.9	36	2.8
Mustelus antarcticus	T	47	2.8	37	2.9	36	2.8
Trygonoptera ovalis	R3	50	2.8	37	2.9	36	2.8
Ophthalmolepis lineolatus	R3	52	2.8	37	2.9	36	2.8
Argyrosomus japonicus	R2	42	2.8	30	5.7		
Scobinichthys granulatus	R3	45	2.8	30	5.7		
*Sepioteuthis australis**	R3	48	2.8	30	5.7		
Achoerodus gouldii	R3	49	2.8	30	5.7		
*Sepia apama**	R2	51	2.8	30	5.7		
Rhabdosargus sarba	R3	53	1.4			36	2.8
Enoplosus armatus	R3	55	1.4			36	2.8
Dactylophora nigricans	R3	56	1.4			36	2.8
Hypoplectrodes nigroruber	R3	57	1.4			36	2.8
Meuschenia venusta	R3	58	1.4			36	2.8
Parupeneus chrysopleuron	R2	54	1.4	37	2.9		
Siganus fuscescens	R3	59	1.4	37	2.9		
Pseudorhombus jenynsii	R3	60	1.4	37	2.9		

3.4.2.2. Univariate diversity and abundance indices

Two-way ANOVA showed that the number of taxa differed significantly between reefs, among months and with the reef×month interaction (Table 3.5a). The reef main effect explained by far the greatest proportion of the variance (76.1%), with month and the interaction term being relatively minor (11.4% and 8.6%, respectively). The mean number of taxa was significantly higher at Dunsborough than Bunbury in each of the months and particularly so in December (14 *vs* 5) and April (16 *vs* 6, respectively; Fig. 3.11). This difference was far less marked in February (14 *vs* 13.5 in Dunsborough and Bunbury, respectively), which explains the relatively minor interaction between reef and month. The number of taxa displayed a general trend of increasing values over spring and summer months, though in Bunbury these values began to decrease after February, whilst in Dunsborough this began after April, with these differences also contributing to the significant interaction (Fig. 3.11).

Table 3.5. Degrees of freedom (df), mean squares (MS), percentage of variance explained by the mean squares (% var), F-values (F) and significance levels (*p*) from a two-way ANOVA on (a) number of taxa, (b) total MaxN, (c) Simpson's Index and the MaxN of (d) *Chrysophorus auratus*, (e) *Pseudocaranx* spp. and (f) *Seriola hippos*, recorded on the Bunbury and Dunsborough artificial reefs between October 2015 and June/July 2016. Significant results are highlighted in bold (p<0.05).

		(a) Number of taxa				(b) Total MaxN			
Factors	**df**	**MS**	**% var**	**F**	***p***	**MS**	**% var**	**F**	***p***
Reef	1	**221.090**	**76.10**	**19.600**	**0.001**	**2.400**	**79.50**	**28.010**	**0.001**
Month	8	**33.130**	**11.40**	**2.940**	**0.009**	**0.200**	**6.60**	**2.310**	**0.033**
Reef×Month	8	**24.970**	**8.60**	**2.210**	**0.041**	**0.330**	**11.00**	**3.890**	**0.001**
Residual	53	11.290	3.90			0.090	2.90		

		(c) Simpson's Index				(d) *Chrysophorus auratus* MaxN			
Factors	**df**	**MS**	**% var**	**F**	***p***	**MS**	**% var**	**F**	***p***
Reef	1	0.010	12.70	0.560	0.459	**1.357**	**75.50**	**7.020**	**0.011**
Month	8	0.020	32.40	1.370	0.233	0.124	6.90	0.640	0.739
Reef×Month	8	0.020	31.00	1.290	0.267	0.124	6.90	0.640	0.739
Residual	53	0.020	24.00			0.193	10.70		

		(e) *Pseudocaranx* spp. MaxN				(f) *Seriola hippos* MaxN			
Factors	**df**	**MS**	**% var**	**F**	***p***	**MS**	**% var**	**F**	***p***
Reef	1	**16.400**	**86.20**	**65.690**	**0.001**	**0.410**	**47.90**	**5.460**	**0.025**
Month	8	**1.250**	**6.60**	**5.020**	**0.001**	**0.240**	**28.10**	**3.190**	**0.008**
Reef×Month	8	**1.110**	**5.90**	**4.450**	**0.001**	0.130	15.10	1.720	0.127
Residual	53	0.250	1.30			0.080	8.80		

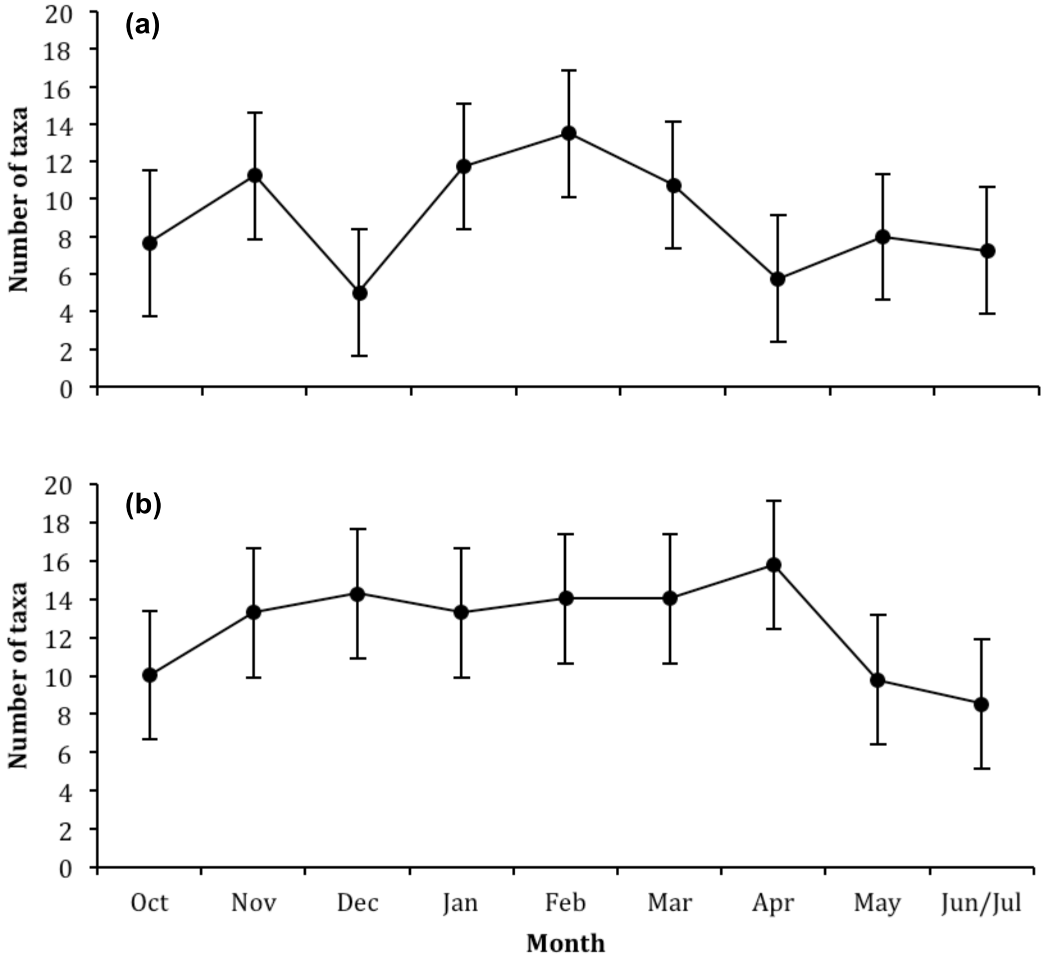


Fig. 3.11. Mean number of taxa recorded on the (a) Bunbury and (b) Dunsborough artificial reefs in each month between October 2015 and June/July 2016. Error bars represent ± 95% confidence intervals.

Significant differences in total MaxN were also recorded between reefs, months and the one-way interaction between these main factors (Table 3.5b). Almost 80% of the variation in this variable was explained by reef, followed by the reef×month interaction and month (11% and 6.6%, respectively). In all months, except October and November, total MaxN was greater on the Dunsborough reef compared to Bunbury (Fig. 3.12). While mean total MaxN was relatively consistent among months at Bunbury ranging from 16 in April to 55 in November, values for this varied far more at Dunsborough (22 in October to 177 in March). Among months at Bunbury, the greater mean total MaxN values were recorded in the spring and summer months, excluding December, compared to the autumn and winter months. At Dunsborough, higher values occurred in summer and autumn (typically >100), compared to winter and spring (<53). These differences in the monthly trends at each reef were, in part, responsible for the significant reef×month interaction.

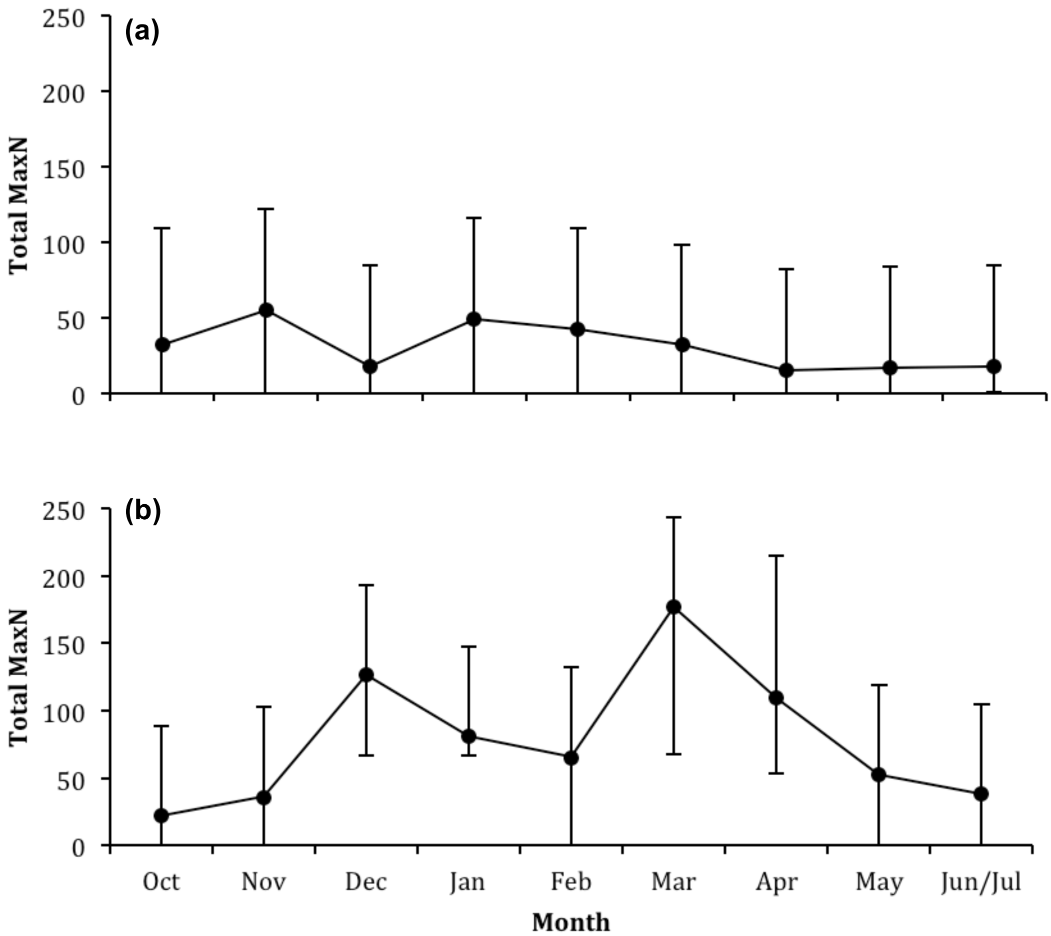


Fig. 3.12. Mean total MaxN recorded on the (a) Bunbury and (b) Dunsborough artificial reefs in each month between October 2015 and June/July 2016. Error bars represent ± 95% confidence intervals.

A two-way ANOVA demonstrated Simpson's Index did not differ significantly among any main effect or interaction (Table 3.5c). Mean values ranged from a low of 0.62 during November at Dunsborough to a high of 0.88 during the following month, on the same reef (data not shown).

The MaxN of *C. auratus* differed significantly among only the reef main effect, which explained 75.5% of the observed variance (Table 3.5d). The average MaxN at Dunsborough was 0.42, while this species was not recorded on the Bunbury artificial reef (Fig. 3.13).

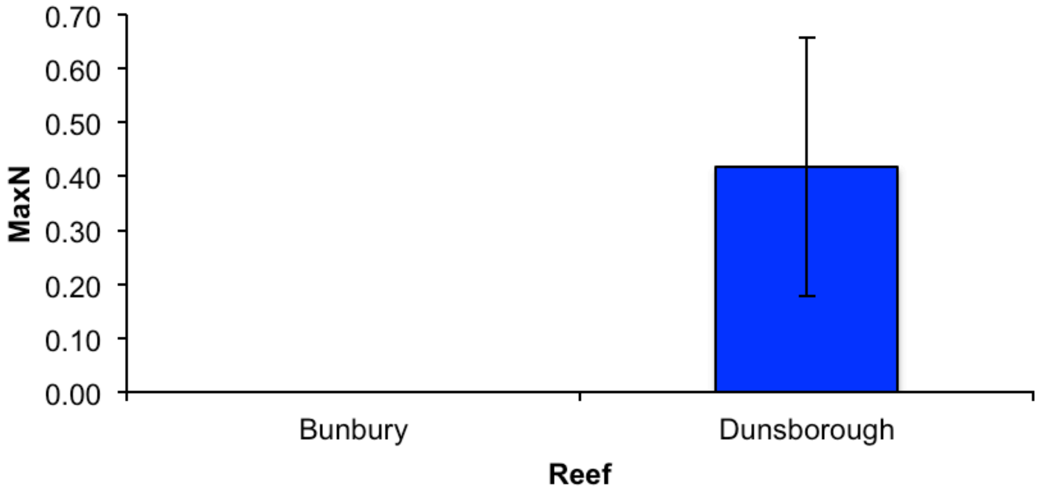


Fig. 3.13. Mean MaxN of *Chrysophorus auratus* recorded on the Bunbury and Dunsborough artificial reefs. Error bars represent ± 95% confidence intervals.

MaxN of *Pseudocaranx* spp. was found to exhibit significant differences with all three factors, *i.e.* reef, month and the reef×month interaction (Table 3.5e). Once again, reef was found to be the most important term in the model, accounting for 86.2% of the variance, with month and the interaction only making minor contributions (6.6% and 5.9%, respectively). In all months except October and November, the MaxN of *Pseudocaranx* spp. was greater on the Dunsborough reef compared to Bunbury (Fig. 3.14). The monthly pattern of abundance differed among reefs, with a pronounced peak occurring at Bunbury in November (16 *vs* 1-3 during the other months), whereas this peak occurred between December and January at Dunsborough (32 and 32, respectively, *vs* 2-13 in the other months). This mismatch in the monthly abundances accounts for the significant result in the reef×month interaction.

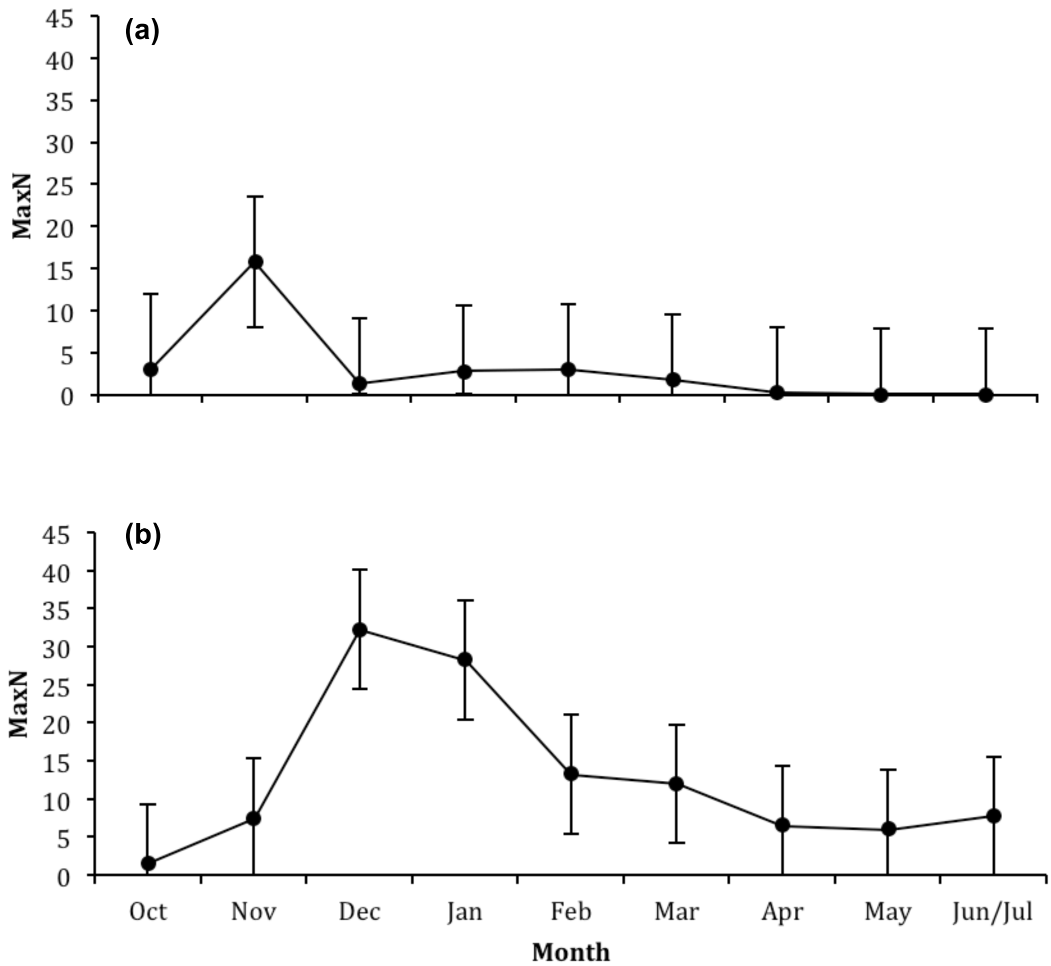


Fig. 3.14. Mean MaxN of *Pseudocaranx* spp. recorded on the (a) Bunbury and (b) Dunsborough artificial reefs in each month between October 2015 and June/July 2016. Error bars represent ± 95% confidence intervals.

A two-way ANOVA detected significant differences in the monthly MaxN of *S. hippos* and among reefs, but not in the interaction between these main effects (Table 3.5f). Reef (47.9%) was also shown to explain a larger proportion of the variance than month (28.1%). The MaxN of *S. hippos* was greater at Dunsborough than Bunbury, *i.e.* 0.387 vs 0.192 individuals per video, respectively (Fig. 3.15). The abundance of *S. hippos* was quite variable with relatively large values in January and June/July (3.9 and 3.6, respectively) and a low of 0.9 individuals per video in April (Fig. 3.16).

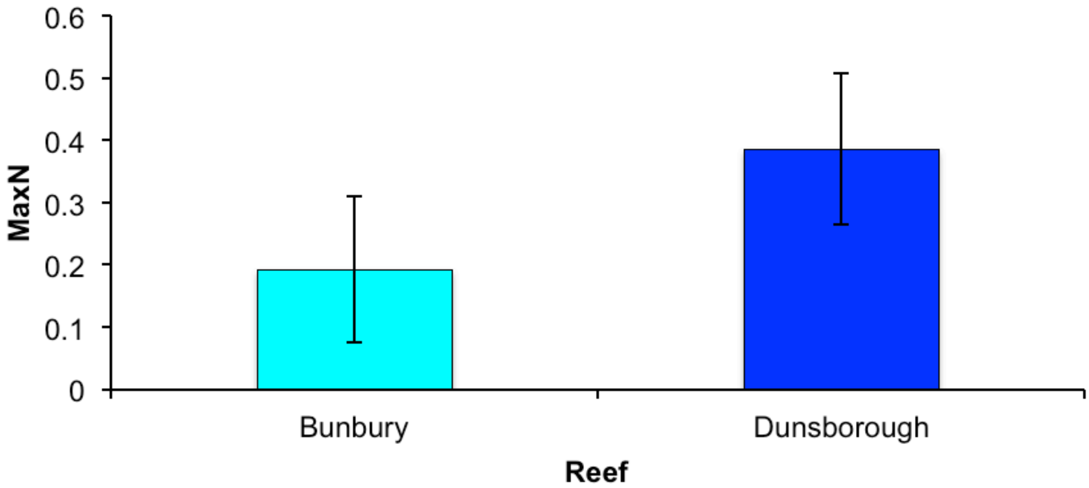


Fig. 3.15. Mean MaxN of *Seriola hippos* recorded on the Bunbury and Dunsborough artificial reefs. Error bars represent ± 95% confidence intervals.

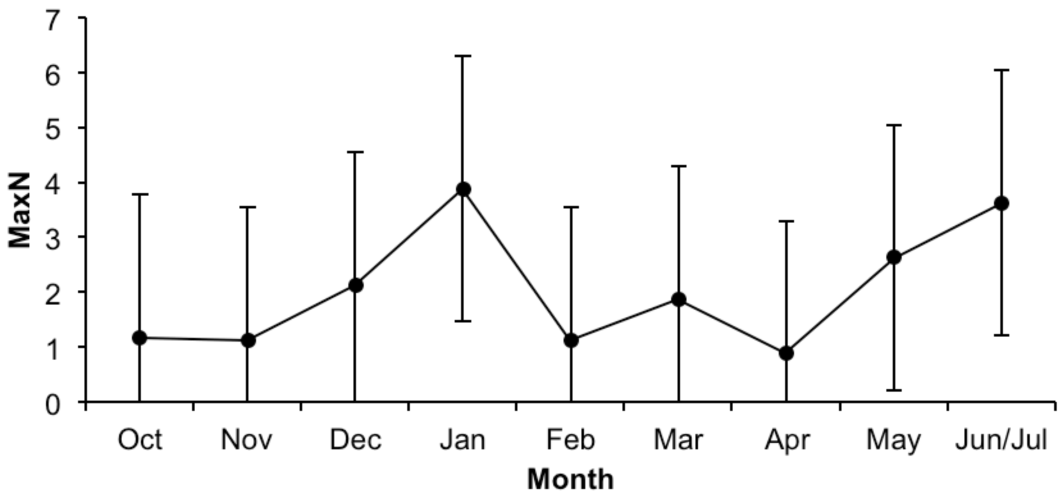


Fig. 3.16. Mean MaxN of *Seriola hippos* recorded on the Bunbury and Dunsborough artificial reefs in each month between October 2015 and June/July 2016. Error bars represent ± 95% confidence intervals.

3.4.2.3. Contribution of taxa and individuals to guilds

Among the 60 taxa recorded, 19 (32%) were classified as transient, meaning they were never present in more than two consecutive months. In contrast, 16 (26%) were classified as resident level 1, i.e. found in ≥ than 87.5% of the months, 15 (25%) were classified as resident level 2, meaning they were found in ≥50% of the months and a further 10 (17%) were resident level 3, i.e. found in two or more consecutive months (Fig. 3.17).

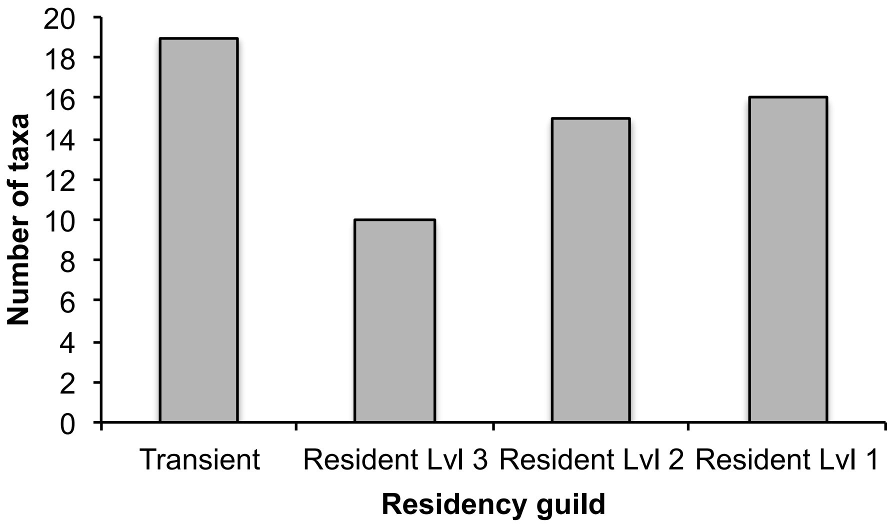


Fig. 3.17. The number of taxa from both reefs in each of the residency guilds.

PERMANOVA found that the percentage contribution of individuals from each of the habitat guilds did not differ significantly among reef, month or the reef×month interaction (Table 3.6a). The overall reef communities were composed of ~23% benthic, ~36% epibenthic and ~41% pelagic taxa.

Table 3.6. Degrees of freedom (df), mean squares (MS), percentage of variance explained by the mean squares (% var), F-values (F) and significance levels (*p*) from a two-way PERMANOVA on the percentage contribution of (a) habitat guilds by individuals, (b) habitat guilds by taxa, (c) feeding guilds by individuals, and (d) feeding guilds by taxa, recorded on the Bunbury and Dunsborough artificial reefs between October 2015 and June/July 2016. Significant results are highlighted in bold ($p \leq 0.05$).

		(a) Habitat guilds by individuals				**(b) Habitat guilds by taxa**			
Factors	**df**	**MS**	**% var**	**F**	***p***	**MS**	**% var**	**F**	***P***
Reef	1	971.05	44.5	1.81	0.17	**514.07**	**6.7**	**3.99**	**0.03**
Month	8	116.13	5.3	0.22	0.99	144.14	1.9	1.12	0.39
Reef×Month	8	559.24	25.6	1.04	0.42	156.37	2.1	1.22	0.29
Residual	53	537.25	24.6			6822.20	89.3		

		(c) Feeding guilds by individuals				**(d) Feeding guilds by taxa**			
Factors	**df**	**MS**	**% var**	**F**	***p***	**MS**	**% var**	**F**	***p***
Reef	1	**1371.43**	**3.8**	**3.34**	**0.04**	407.33	38.9	2.42	0.09
Month	8	366.34	1.0	0.89	0.56	207.98	19.9	1.23	0.25
Reef×Month	8	**873.88**	**2.5**	**2.13**	**0.02**	264.53	25.2	1.58	0.10
Residual	53	33223.55	92.7			168.01	16.0		

When each species was allocated to a habitat guild, PERMANOVA found that the percentage of species in each guild differed significantly for the factor reef, although it should be noted that this factor only explained 6.7% of the variance observed, compared to 89% in the residual (Table 3.6b). Differences in guilds between the reefs were due to the fact that Dunsborough had a higher mean percentage contribution of benthic taxa than Bunbury (32.8% *vs* 23.9%, respectively), while the reverse was true for pelagic taxa (32.6% *vs* 44.1%, respectively; Fig. 3.18).

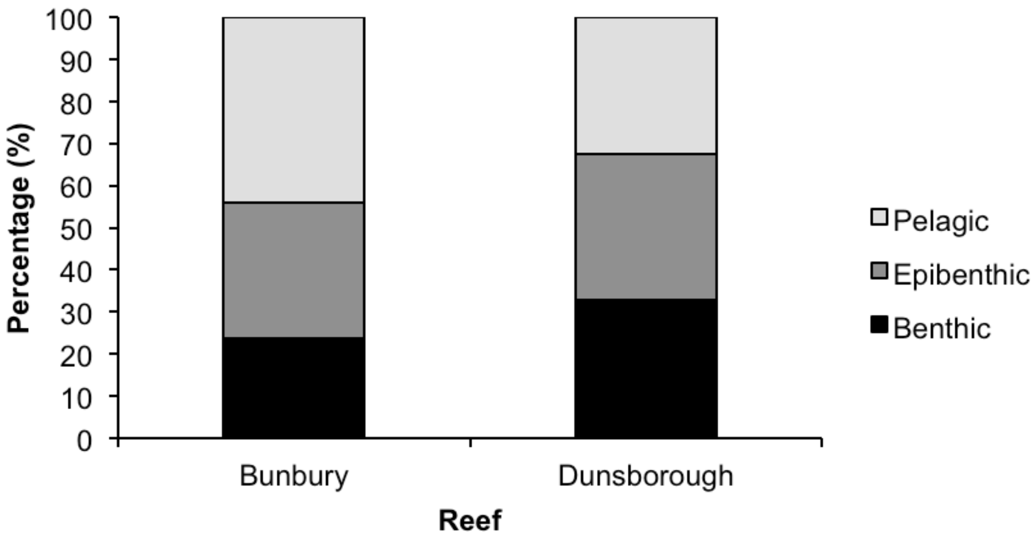


Fig. 3.18. Percentage contribution of taxa to the various habitat guilds at the Bunbury and Dunsborough artificial reefs.

When testing percentage composition of feeding guilds based upon contribution by individuals, significant differences were detected for the reef main effect and the reef×month interaction (Table 3.6c). However, these factors explained relatively small proportions of the variance, with reef contributing 3.8% and the interaction explaining 2.5%. When comparing reefs Bunbury typically had a greater proportion of zooplanktivores and a lower contribution of zoobenthivores, although the contributions were quite variable among months, thus helping to explain the reef×month interaction. The percentage contribution of the remaining guilds was relatively minor and typically variable, although herbivores were exclusively encountered on the Bunbury reef, albeit only in January (Fig. 3.19).

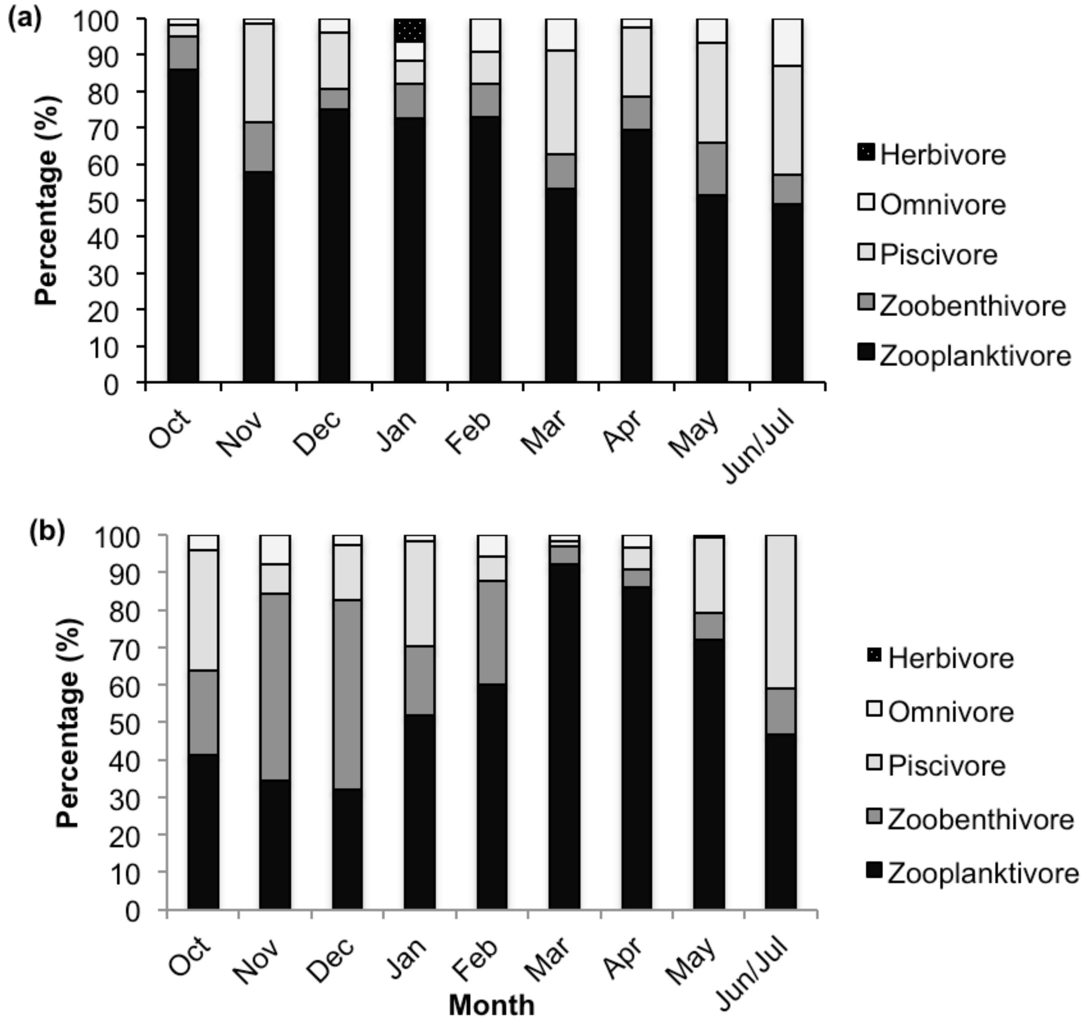


Fig. 3.19. Percentage contribution of individuals to the various feeding guilds at the (a) Bunbury and (b) Dunsborough artificial reefs in each month between October 2015 and June/July 2016.

PERMANOVA demonstrated that the percentage number of species in each feeding guild, did not differ among any of the terms in the model (Table 3.6d). The mean percentage composition of each guild over both reefs was as follows, zooplanktivore ~32%, zoobenthivore ~24%, piscivore ~27%, omnivore ~15% and herbivore ~2%.

3.4.2.4. Fish community composition

A two-way PERMANOVA on the fish community composition detected significant differences among reef, month and the reef×month interaction (Table 3.7). The reef main effect explained around four times more of the variation than either month or the reef×month interaction.

Table 3.7. Degrees of freedom (df), Mean Squares (MS), percentage of variance explained by the means squares (% var), Pseudo-F values (F) and significance levels (*p*) from a PERMANOVA on the fish community composition of the Bunbury and Dunsborough artificial reefs in each month between October 2015 and June/July 2016. Significant results are highlighted in bold ($p<0.05$).

Factors	df	MS	% var	F	*p*
Reef	1	**11363**	**60.7**	**6.93**	**0.001**
Month	8	**2814**	**15.0**	**1.72**	**0.002**
Reef×Month	8	**2917**	**15.6**	**1.78**	**0.001**
Residual	53	1639	8.7		

The nMDS ordination plot shows a distinct demarcation between reefs, with the point representing the Bunbury reef occurring predominantly to the left of those for the Dunsborough reef. The points for both reefs also show a general clockwise cycling pattern starting from October through to June/July. The interaction between the factors is also apparent with the reefs showing differences in the distances between their monthly samples. For example, the months December and January are located relatively far apart from one another for Bunbury, but are tightly coupled for Dunsborough. This pattern is also repeated with March and April, though to a lesser extent (Fig. 3.20).

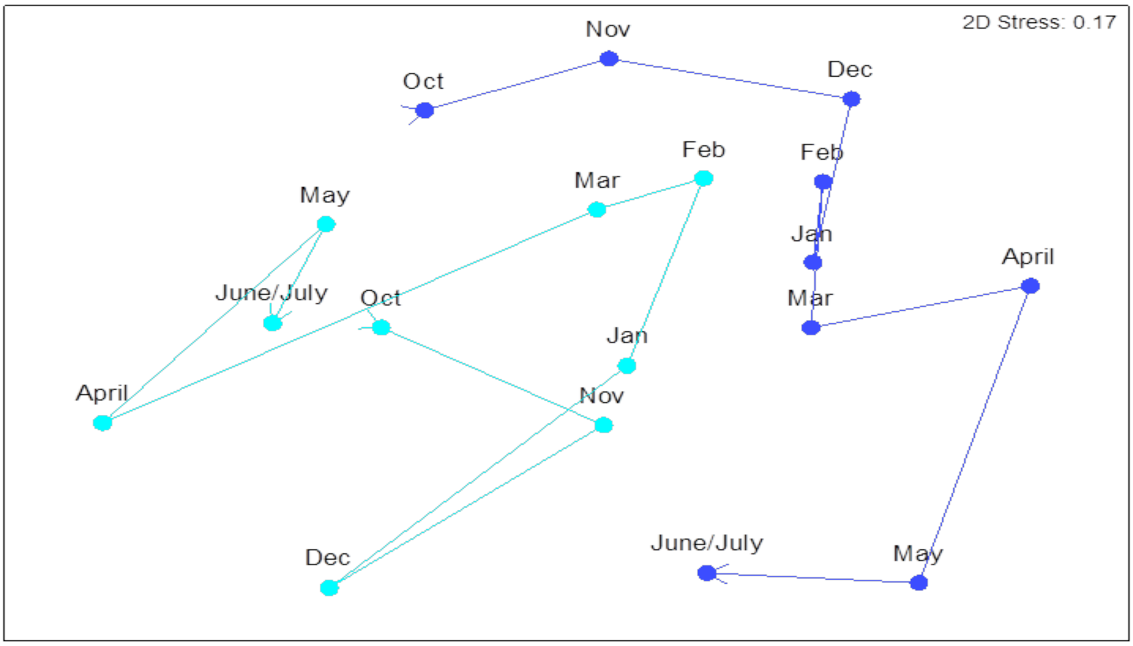


Fig. 3.20. nMDS ordination plot, derived from separate Bray-Curtis similarity matrices, constructed from the square-root transformed and averaged MaxN fish community data recorded on the ●Bunbury and ● Dunsborough artificial reefs between October 2015 and June/July 2016. Separate trajectories for each reef have been plotted to allow tracking of the fish composition of each reef through time.

Among the two artificial reefs, pairwise comparisons generated by PERMANOVA, demonstrated that fish faunal composition differed significantly in November and December, as well as March through May. The biggest differences according to the magnitude of the t-values, were for April, followed closely by May (Table 3.8). From the shade plot it is apparent that the presence of some key species at the Dunsborough reef, and absence or low abundance at the Bunbury reef likely made a large contribution to the differences in overall assemblage seen between reefs. These taxa include the Crested Morwong *Cheilodactylus gibbosus,* Globefish *Diodon nicthemerus,* Spotted Grubfish *Parapercis ramsayi*, Wavy Grubfish *Parapercis haackei, C. auratus,* Bluethroat Rainbow Wrasse *Suezichthys cyanolaemus* and the Moonlighter *Tilodon sexfasciatus.* While the exclusivity, or majority presence, of the West Australian Butterflyfish *Chaetodon assarius*, Longspine Flathead *Platycephalus* *longispinis,* Snakeskin Wrasse *Eupetrichthys angustipes* and the Masked Stingaree *Trygonoptera personata* on the Bunbury reef also likely made a large contribution to the observed differences. Specifically, differences observed between the reefs in April, which had the largest t-value, were driven by *C. auratus*, *C. gibbosus*, *P. haackei* being recorded exclusively on the Bunbury reef. Additionally, Rough Bullseye *Pempheris klunzingeri*, *C. assarius* and *P. longispinis* were noted as absent on the Dunsborough reef, but were recorded on the Bunbury reef (Fig. 3.21).

Table 3.8. T-values derived from a pairwise PERMANOVA test on the reef×month interaction, demonstrating the months in which the fish composition recorded on the Bunbury and Dunsborough artificial reefs differed. Significant pairwise comparisons are highlighted in grey ($p<0.05$).

Month	t-value
Oct	1.287
Nov	1.359
Dec	1.627
Jan	1.117
Feb	1.065
Mar	1.615
Apr	2.074
May	2.068
Jun/Jul	1.333

A pairwise PERMANOVA test between months for each reef revealed that the composition of the fish fauna at Bunbury differed only between April *vs* November, February and March (Table 3.9a). This was due to fewer species being caught during April, with species, such as the Southern Fiddler Ray *Trygonorrhina dumerilli*, *M. australis, C. curiosus* and *N. obliquus,* all absent, but recorded in relatively high abundances in most months (Fig. 3.21). Thus, it was the depauperate nature of April that made it unique. Dunsborough exhibited a greater number of monthly differences, with 20 of the 36 pairwise comparisons being significant and typically only successive months not differing (Table 3.9b). Fish faunal composition was fairly consistent in terms of the presence/absence of species and thus the significant differences are driven by changes in the relative abundance of species such as *Pseudocaranx* spp., *N. obliquus*, *A. amygdaloides* and *S. hippos* (Fig. 3.21).

Table 3.9. T-values derived from a pairwise PERMANOVA test on the reef×month interaction, demonstrating the months in which the fish composition recorded on the (a) Bunbury and (b) Dunsborough artificial reefs differed. Significant pairwise comparisons are highlighted in grey (p<0.05).

(a) Month	Oct	Nov	Dec	Jan	Feb	Mar	Apr	May
Nov	1.16							
Dec	1.01	0.97						
Jan	0.99	0.88	0.96					
Feb	1.14	1.43	1.23	1.06				
Mar	1.41	1.23	1.26	1.05	1.24			
Apr	1.18	1.59	0.92	1.31	1.67	1.70		
May	1.05	1.45	1.14	1.22	1.44	1.33	0.79	
Jun/Jul	0.81	1.34	1.2	1.04	1.36	1.30	0.97	0.70

(b) Month	Oct	Nov	Dec	Jan	Feb	Mar	Apr	May
Nov	1.24							
Dec	1.62	1.35						
Jan	1.68	1.45	1.13					
Feb	1.47	1.38	0.98	1.16				
Mar	1.63	1.53	1.41	1.22	0.78			
Apr	2.21	2.09	1.76	1.57	0.97	0.71		
May	2.16	2.11	1.86	1.42	1.8	1.65	1.86	
Jun/Jul	1.42	1.38	1.48	1.44	1.49	1.32	1.62	1.17

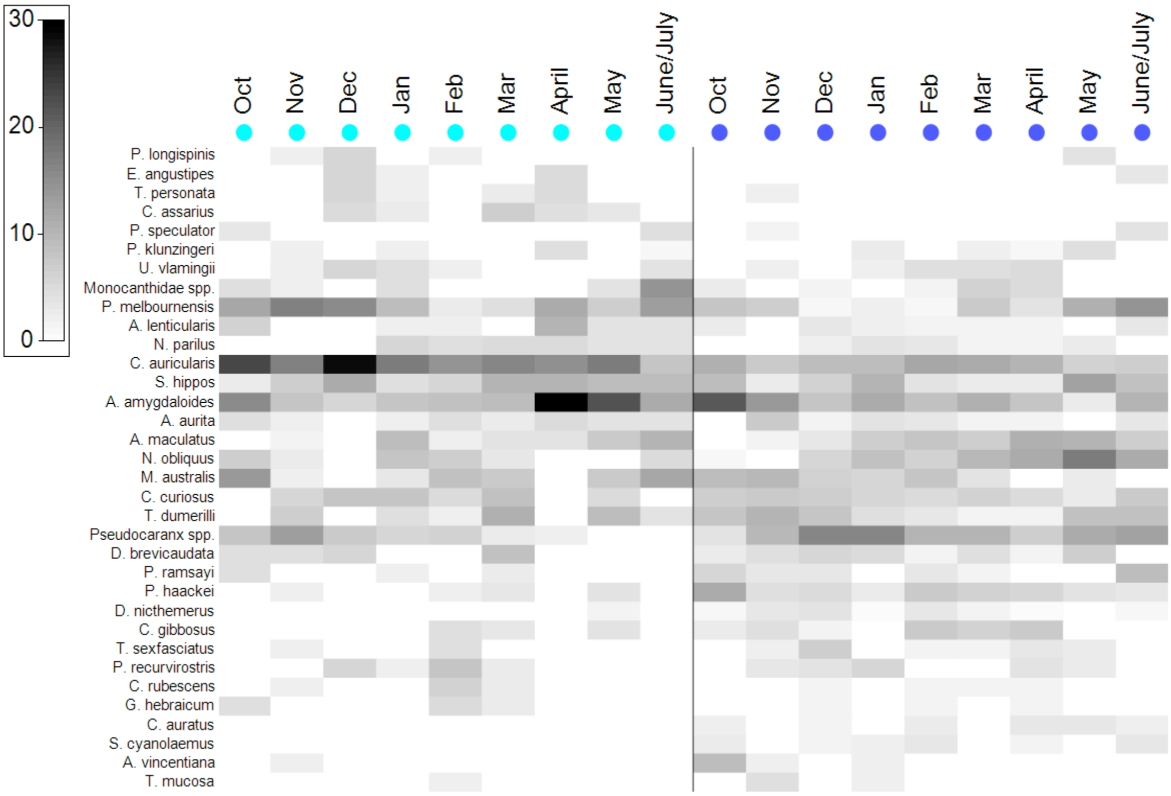


Fig. 3.21. Shade plot, constructed using the square-root transformed and averaged MaxN abundances of each taxon recorded on video footage recorded on the ● Bunbury and ● Dunsborough artificial reefs in each month between October 2015 and June/July 2016. White space denotes the absence of a taxon, with the grey scale representing the transformed and averaged MaxN abundances. Note only species that occurred in ≥5% of the videos from either reef are included.

The segmented bubble plot of key recreational species on the Bunbury artificial reef shows that *S. hippos* was found at a relatively similar abundance in every month, whereas the West Australian Dhufish *Glaucosoma hebraicum* was found only between October and March, although not in every month. Other species, such as *Pseudocaranx* spp. and the Baldchin Groper *Choerodon rubescens,* appeared sporadically (Fig. 3.22a). Although *Pseudocaranx* spp. was found in every month at Dunsborough, the abundances of other species, such as *G. hebraicum*, *S. hippos* and *C. rubescens* were typically restricted to months between January and April (Fig. 3.22b).

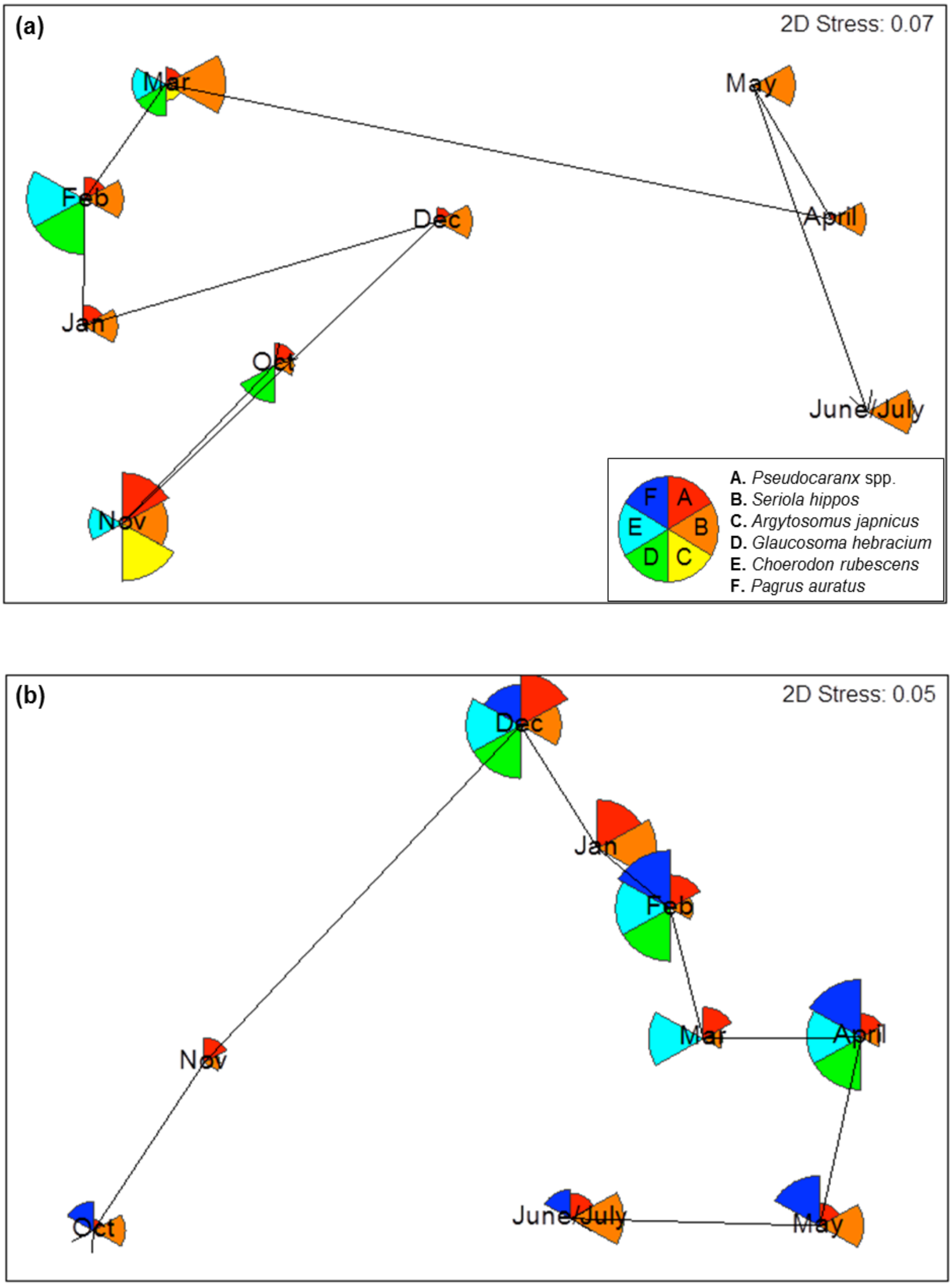


Fig. 3.22. nMDS ordination plots, derived from separate Bray-Curtis similarity matrices, constructed from the square-root transformed and averaged MaxN fish community data recorded on the (a) ●Bunbury and (b) ● Dunsborough artificial reefs in each month between October 2015 and June/July 2016. Segmented bubbles of proportional sizes are overlaid illustrating the relative abundance of six key recreational target species.

3.5. Discussion

This research project utilised BRUV technology, in conjunction with a citizen science based sampling program (Reef Vision), to compare spatial and temporal trends in the fish communities of the artificial reefs of Geographe Bay (*i.e.* Bunbury and Dunsborough) between October 2015 and July 2016 inclusive. A range of statistical tests were undertaken to elucidate whether the number of taxa, total MaxN, the MaxN of a number of recreationally important fish species, fish community composition and the contributions of habitat and feeding guilds differed spatially, *i.e.* between the Bunbury and Dunsborough reefs, and temporally, *i.e.* in months between October and July. The following sections discuss the patterns of variation in (i) the number of taxa and total abundance, (ii) the proportions of species and individuals to the different residency, habitat and feeding guilds and (iii) the faunal community structure, and detail how and why these characteristics changed either spatially and/or temporally.

3.5.1. Number of taxa and total MaxN

3.5.1.1. Differences among reefs

Significant differences in the number of taxa and total MaxN were detected between reef, month and their interaction. The reef main effect accounted for by far the greatest proportion of the variance observed, with the values recorded at Dunsborough typically being larger than the corresponding values at Bunbury. This finding mirrors the work by Bateman (2015), who also found significant differences in the fish communities of the same artificial reefs, in terms of species richness, abundance and composition, with the Dunsborough reef supporting a higher richness and abundance of fish than Bunbury. Despite the relatively small distance between the two artificial reefs (~50 km), their differences in the number of taxa and the abundance of key species could potentially be attributed to their varying levels of connectivity with nearby natural reefs. Distance between reefs is believed to be an important determinant of associated fish communities, with studies finding both an increase (Gascon and Miller, 1981; Walsh, 1985; Bombace et al., 1994) and decrease (Molles, 1978; Bohnsack, 1979; Gladfelter et al., 1980) of diversity and abundance with the factor.

In order to understand how the results from this study fit in with these findings, it is necessary to determine how fish communities utilise natural reefs. Temperate limestone reefs in Western Australia have been found to support higher abundances of fish than either seagrass meadows or bare sand habitats (Howard, 1989), with the increased structural complexity of reef habitats thought to be responsible (Borges-Souza et al., 2013; Graham and Nash, 2013). Moreover, Molles (1978) and Harman and Kendrick (2003) found that natural reefs with higher vertical profiles supported greater numbers of fish species and individuals, which has also been documented on artificial reefs (Strelcheck et al., 2005).

It has been suggested that differences in the diversity and abundance of fish communities between habitats are driven by behavioural preferences. For example, individuals may move to areas with lower population densities to reduce intra- and interspecific competition and predation (Ault and Johnson, 1998). This could potentially explain the observed increase in species richness and abundance with increasing isolation of artificial reefs (Gascon and Miller, 1981; Walsh, 1985; Bombace et al., 1994). However, this phenomenon may also be scale dependent, as some studies that cited this were comparing reefs over much smaller distances than in the current study (Gascon and Miller, 1981; Walsh, 1985).

Despite the preference for fish to move to more isolated habitats, with lower population densities, they are limited in their ability to do so by their vagility. Molles (1978) suggested that pelagic fish with a high vagility may be able to traverse relatively large distances between reefs better than less mobile reef-associated fish. Pelagic species are more able to freely move between habitats, regardless of their isolation, while species with a lower vagility, *e.g.* benthic and epibenthic species, are less able to migrate to isolated habitats. This assertion is supported by Vega Fernández et al. (2007), who found higher fish abundances, particularly of smaller less mobile fish species, on artificial reefs with high reef connectivity, whilst larger mobile species were more likely to be associated with isolated artificial reefs.

The south-west edge of Geographe Bay has a high level of reef connectivity, due to the number of limestone and granite reefs occurring near Cape Naturaliste (Westera et al., 2007; Department of Environment and Conservation, 2013), which are thought to influence nearby fish communities (Westera et al., 2007). The results of a BRUV-based study found the abundance of some reef-associated species to decline with increasing distance from Cape Naturaliste, suggesting sparser reef coverage in the bay moving northerly. Therefore, the Dunsborough artificial reef is likely to have a higher connectivity with natural reefs, than the Bunbury artificial reef. This connectivity of the Dunsborough artificial reef, as opposed to Bunbury's relative isolation, is likely to have led to its higher observed number of taxa and total MaxN. This is due to the ability and preference of fish, regardless of their vagility, to move between the connected natural reefs and the Dunsborough artificial reef, either temporarily or permanently.

Vagility and habitat connectivity provide possible explanations for the presence and movement of juvenile and adult fish, however, they do not take into consideration larval dispersal. The dispersal of larval propagules is likely to overcome any boundaries present between habitats on small to medium scales, particularly in Geographe Bay where particle retention rates (residence times) are high (Cowen et al., 2006; Feng et al., 2010). With larval propagule dispersal showing variability both between and within species, as well as over space and time, it is difficult to estimate the impact it has upon fish communities as a whole (Kinlan et al., 2005). It is likely that the abundance of reef habitat in the proximity of the Dunsborough reef would make this area more suitable for reef fish communities, resulting in higher survival rates of fish from the larval stage. Therefore, despite larval dispersal overcoming gaps in habitat connectivity present in Geographe Bay, the presence of more suitable habitat around the Dunsborough reef likely contributed to higher survival rates and therefore, a greater abundance and richness of fish at the reef, compared to Bunbury.

Another factor that could have contributed to differences in number of taxa and abundance observed between the reefs could be depth, as the Bunbury and Dunsborough reefs are situated at depths of 17m and 27m, respectively (Government of Western Australia, 2013). Environmental factors, such as water temperature, light and hydrodynamics, are known to change with depth, thus affecting the growth and condition of fish species (Bayle-Sempere et al., 1994). However, given the small difference in depth between the sites and the ability of some species to tolerate a wide depth range, it is unlikely that increasing depth has contributed to large-scale differences in their fish communities (Srinivasan, 2003). Rather, it is more likely to have species-specific impacts, particularly for taxa that can only tolerate a small depth range.

3.5.1.2. Differences among months

The significant difference observed among months, for both the number of taxa and total MaxN, could potentially be attributed to the impact of changing water temperature with season. A number of studies of artificial reefs have found that water temperature influences associated fish communities (Hastings et al., 1976; Sanders Jr et al., 1985; Stephens Jr et al., 1994; Fujita et al., 1996; Bortone et al., 1997). Seasonal changes in water temperature are used as cues for both migration and reproduction, with gonadal development in many fishes being temperature dependent (Ware and Tanasichuk, 1989). Specifically, fish in temperate marine environments generally release their larvae during spring or autumn (Cushing, 1990; Brodersen et al., 2011). A number of studies on artificial reefs have noted a pattern of high fish abundance during the warmer summer and autumn months, compared to the cooler winter and spring months (Hastings et al., 1976; Bohnsack et al., 1994; Relini et al., 1994; Fujita et al., 1996). These differences in abundance have been attributed to the arrival of juveniles during summer (Bohnsack et al., 1994; Relini et al., 1994); however, Burt et al. (2009) found the majority of the change in abundance observed in their study was driven by the arrival of adults. The movement of adults between seasons is primarily due to migrational behaviour to spawn and/or access increased food availability (Hastings et al., 1976; Bohnsack et al., 1994; Fujita et al., 1996; Burt et al., 2009). Thus, the generally higher values of the number of taxa and total MaxN during spring and summer, observed over both reefs, can be attributed to the movement of both juveniles and adults onto the reefs, triggered by changing sea surface temperatures in Geographe Bay (McMahon et al., 1997).

The process by which communities change over time, ecological succession, could also potentially contribute to the differences observed between months. The theory of ecological succession postulates that communities of new, or disturbed, habitats are not static, rather they constantly change until they reach a point of stabilisation (Anderson, 2007). Bohnsack (1991) found artificial reefs to take between one and five years for climax communities to develop depending upon a range of environmental factors. Thus, since the reefs have only been deployed for just over three years, their associated fish communities are potentially still undergoing successional changes. These changes could contribute to differences observed over time, however, given that the monitoring only took place over only ten months it would be difficult to attribute significant monthly differences completely to this.

3.5.1.3. Difference in monthly variation between locations

Like month, the reef×month interaction was significant for the number of taxa and total MaxN, though had a lesser mean squares than the factors reef or month. This effect is likely due to the variability of oceanographic processes between sites, as well as with time. Oceanographic processes are known to play an important role in determining food availability, ultimately helping to shape ecological communities (Vanni, 1987; Mann, 1993). With the considerable variability oceanographic processes show, over both spatial and temporal scales, the artificial reefs are likely to reflect these differences in their fish communities (Taylor et al., 2013).

Between October and April, the poleward flowing Leeuwin Current weakens due to the influence of northerly winds and the mass movement of water northward via the Capes Current (Gersbach et al., 1999; Department of

Environment and Water Resources, 2006). The latter current promotes the equatorward movement of cool water along the inshore of the south-west coast, causing localised upwellings and enhanced nutrient availability, stimulating algal blooms and attracting aggregations of fish to the area (Gersbach et al., 1999; Pearce and Pattiaratchi, 1999; Department of Environment and Water Resources, 2006). Thus, it would be predicted that areas influenced by this current would temporarily experience a greater number of fish species and individuals due to increased food availability over summer, which as a general trend is borne out in this study. Differences within the patterns of diversity and abundance at each reef could be explained by the differential strength of the current, with Dunsborough being located closer to its strongest point (between Cape Leeuwin and Cape Naturaliste), and thus being impacted for a longer period of time than the Bunbury reef (Pearce and Pattiaratchi, 1999). This resulted in the continuation of high numbers of taxa and total MaxN into autumn at Dunsborough, whereas Bunbury began to experience a decrease in these variables during summer.

3.5.1.4. Differences in the abundance of recreationally important species

Given the purpose of the Bunbury and Dunsborough artificial reefs was to increase the abundance of specific targeted recreational species, *i.e.* *C. auratus*, *Pseudocaranx* spp. and *S. hippos* (Department of Fisheries, 2016), it is important to understand the factors influencing their distribution. The first of these, *C. auratus*, is found along the coastline of southern Australia, predominantly occurring on rocky reef to depths of up to 200 m (Moran et al., 1999; Fowler et al., 2004; Edgar, 2008; Parsons et al., 2014). The higher abundance of this species on the Dunsborough reef, compared to Bunbury, is thought to be related to the fact that the former is located in closer proximity to natural reefs. However, it is unlikely that there is a complete lack of individuals on the Bunbury reef, as observed in the current study, as local recreational fishers have noted catching the species in the area (Reef Vision participants 2016, pers. comm., 9 September; Westera et al., 2007; Department of Environment and Conservation, 2013). This recorded absence could be attributed to a variety of factors, including missed observations due to environmental conditions, individuals being recorded on footage that was not analysed and/or a failure in the methodology to record them.

The abundance of *Pseudocaranx* spp. (likely *Pseudocaranx dentex, georgianu*s and *wrighti)* differed both spatially and temporally, with typically greater values recorded on the Dunsborough reef, and both reefs displaying increased MaxN values in spring/summer. Trends in their abundance with reef, month and their interaction are likely driven by the aforementioned factors of habitat connectivity, changes in water temperature and the influence of currents, as their abundance showed similar patterns to those seen overall (Smith-Vainz and Jelks, 2006).

Habitat connectivity is also considered responsible for the increased abundance of *S. hippos* at Dunsborough as this species typically inhabits inshore waters with structured habitat, such as reefs (Rowland, 2009). This species spawns in late spring and throughout summer in Western Australia, with individuals forming spawning aggregations to the west of Rottnest Island (Rowland, 2009). Their observed pattern of abundance on the Geographe Bay artificial reef, *i.e.* typically being low over summer and increasing in late autumn/winter, could be explained by their migration away from the reefs to spawn.

3.5.2. Residency, habitat and feeding guilds

A number of studies on artificial reefs have found wide-ranging results when assigning fish to residency guilds. For example, the percentage of species classified as non-residents ranges between 35% and 90% depending on the study (Relini et al., 1994; Stephens Jr et al., 1994; Santos et al., 2005; Gül et al., 2011). These wide ranging results, particularly when compared to this study, show that without a clear definition of what constitutes 'resident' or 'transient' species comparisons between studies are difficult, even when using a tiered system as employed here (Boisnier et al., 2010). Therefore, no meaningful conclusion can be drawn as to how the fish community use the Geographe Bay artificial reefs, in terms of residency, compared to other deployments.

Differences in the proportion of taxa representing the various habitat guilds, provides more evidence to support the assertion that habitat connectivity plays a key role in determining the fish assemblages present on the artificial reefs. Due to the isolation of the Bunbury reef, it would be more difficult for species with low vagility to immigrate there, thus explaining why highly mobile, pelagic taxa predominate. The closer location of the Dunsborough reef to natural reefs has meant that benthic taxa with low vagility, are able to move between the habitats and hence, make up a greater proportion of the total population.

When classifying individual fish into feeding guilds, significant reef and reefxmonth differences were detected, with the former explaining a far greater proportion of the variance. When comparing reefs, greater proportions of zooplanktivores and zoobenthivores were found on the Bunbury and Dunsborough reefs, respectively. The greater abundance of zooplanktivores in Bunbury is likely related to the presence of strong currents year round. South of Perth, the Leeuwin Current moves into an offshore position, except for an arm that penetrates into Geographe Bay, in the general proximity of Bunbury. This, and the presence of the Capes Current during summer, means that the Bunbury reef is likely to be affected by relatively high current speeds year round, while the Dunsborough reef is likely only affected by large scale currents when the Capes Current moves in during summer (Cresswell and Vaudrey, 1977; Pearce and Pattiaratchi, 1999; Smale, 2012). With the presence of these currents, zooplanktivorous fish are able to increase their feeding rates, making the Bunbury reef an advantageous location to inhabit year round, while the Dunsborough reef is a beneficial site only between late spring and early autumn (Stevenson, 1972; de Boer, 1978; Gersbach et al., 1999; Pearce and Pattiaratchi, 1999). These observations are reflected in the reef×month interaction results, with the Dunsborough reef showing a general increase in the proportion of zooplanktivores in summer and early autumn, while Bunbury shows a more consistent proportion of zooplanktivores year round.

The difference in the proportion of zoobenthivores between the two reefs could be due to the greater provision of structured habitat surrounding the Dunsborough reef. In their work, comparing invertebrate abundances between habitats, Nakamura and Sano (2005) found a greater biomass of invertebrates in reef habitats compared to seagrass meadows, due to the tendency for larger invertebrates to occupy reefs. With the Dunsborough reef likely having greater connectivity with natural reefs, this area may support a greater biomass of invertebrates for zoobenthivores to feed upon, leading to them making up a larger proportion of the community compared to Bunbury (Westera et al., 2007; Department of Environment and Conservation, 2013).

3.5.3. Fish community composition

The significant differences detected in the composition of the fish fauna among reef, month and the reefxmonth interaction is likely due to a variety of factors. These include the aforementioned changes in the overall number of taxa and total MaxN, as well as differences in the abundance of key recreational species and the composition of habitat and feeding guilds.

In an attempt to explain how the overall fish communities differ, pairwise PERMANOVA tests were carried out between months on each reef, and between reefs with month. Beyond the general contribution of the previously covered univariate measures, the shade plot and pairwise tests revealed a number of different fish as having contributed to these differences. Some of these highlighted species were found to occur exclusively, or primarily, on the Dunsborough reef, in a sporadic fashion, including *C. gibbosus, D. nicthemerus, P. ramsayi*, *P. haackei, C. auratus, S. cyanolaemus* and *T. sexfasciatus*. Their abundance at the Dunsborough reef can likely be attributed to their preference for structured habitat, particularly reefs, which have a greater coverage in the vicinity of the site (Hutchins and Thompson, 1983; Moran et al., 1999; Fowler et al., 2004; Westera et al., 2007; Edgar, 2008; Department of Environment and Conservation, 2013; Parsons et al., 2014). However, their absence at the Bunbury reef could also be contributed to a failure in the method to record their occurrence. This is likely to have occurred with *S. cyanolaemus, P. ramsayi* and *P. haackei* due to their colouration and small size, which makes identification difficult, as well as the schooling behaviour of *D. nicthemerus*, which could have caused the patchy abundance recorded (Florisson 2016, pers. comm., 20 April; Edgar, 2008).

The distribution of *C. gibbosus* primarily on the Dunsborough reef appears anomalous, as they are known to occur over both reef and sand habitat, meaning habitat connectivity cannot explain their distribution (Hutchins and Thompson, 1983). This species has been little studied but others in the genus *Cheilodactylus* are noted to occur in patchy aggregations or display territorial behaviour, limiting their densities (McCormick, 1989; Lowry and Suthers, 1998). Thus, the recorded distribution of *C. gibbosus* could potentially be attributed to the low densities in which it occurs, causing missed observations (Edgar, 2008). Furthermore, the zero recorded abundance of *C. auratus* at the Bunbury reef found in this study, as noted earlier, is in contrast to their recorded occurrence in the vicinity of the reef and highlights the potential for the method to fail to record some species (Reef Vision participants 2016, pers. comm., 9 September). Furthermore, the observed differences in water clarity between the sites, over winter could impact the ability to observe small and cryptic species. Bunbury experienced a considerable reduction in water clarity over winter, while Dunsborough also experienced this, but for a shorter duration.

A number of other species displayed the opposite trend, occurring exclusively, or primarily, on the Bunbury reef. These included *C. assarius*, *P. longispinis, E. angustipes* and *T. personata.* The recorded absence of *C. assarius* at the Dunsborough reef is likely linked to its infrequent occurrence on the south-west coast of Australia, as it is found in a greater abundance at higher latitudes within its home range (Hutchins and Thompson, 1983). The abundance of *E. angustipes* and *P. longispinis* at the Bunbury reef could be due to its preference for sandy habitats, which are likely to cover a greater area near the Bunbury reef (Hutchins and Thompson, 1983; Edgar, 2008). Though the small size, colouring and cryptic behaviour of *E. angustipes*

(Gomon et al., 2008) may have caused missed observations on the Dunsborough reef. The greater abundance of *T. personata* at the Bunbury reef is difficult to explain as the species has been found to occur in low densities typical at the depth of this reef (Platell et al., 1998). However, this species has been little studied and its distribution is yet to be fully understood.

The centroid plot, showing the occurrence of the six most abundant recreationally targeted species, reveals that each reef supports a different suite of recreational species. While the patterns of abundance of *S. hippos, Pseudocaranx* spp. and *C. auratus* are discussed above, the appearance of the Western Australian Dhufish *Glaucosoma hebraicum* and Baldchin Groper *Choerodon rubescens* between November and April on the reefs warrants discussion. Mature *G. hebraicum* are known to spawn on reefs between December and March (Hesp et al., 2002; Edgar, 2008). Its appearance on the reefs during these months could be a result of its migration to the area to spawn, while the presence of *C. rubescens* is likely linked to the appearance of new recruits, which generally appear in reef and sand habitats during late summer (Hutchins and Thompson, 1983; Fairclough, 2005).

The characteristics of the fish fauna of the artificial reefs of Geographe Bay differ, both spatially and temporally. Habitat connectivity with adjacent natural reef systems is likely to be largely responsible for structuring the fish communities of the artificial reefs. However, both water temperature and the influence of the Leeuwin and Capes Currents are likely to play a subsidiary, but significant role.

3.5.4. Study limitations and scope for future work

Having evaluated a range of methods for monitoring the faunas of artificial reefs (Chapter 2), this section firstly provides an evaluation of the BRUV methodology and its application in the current study. Secondly, it details ideas for future research on the Geographe Bay artificial reefs, as well as the application of the method to monitor the fauna of other artificial reef deployments.

3.5.4.1. Critical evaluation of the use of BRUVs

Although based on limited data, the results of Florisson (2015) indicated that the placement of the BRUV, *i.e.* facing towards or away from the artificial reef modules, has the potential to influence the types of fish recorded. However, Bateman (2015), using a more comprehensive data set found this factor did not significantly impact the composition recorded. Moreover, the distance of the camera from the reef module likely influences the types of species recorded. Thus, on the few occasions that cameras were placed within the module itself, species not previously observed, such as *Apogon victoriae* and Pempherididae spp., were identified. The identification of cryptic species, such as these, has been noted to show high levels of variability in video footage (Williams et al., 2006). Therefore, the positioning of the BRUV units on the reefs could impact the fish community composition recorded and thus needs to be controlled.

The introduction of sampling bias, both temporally and spatially, by volunteers has been noted as a significant risk within citizen science based monitoring programs (Dickinson et al., 2010). Such biases could have been introduced into this study as sampling dates were not kept consistent and sampling locations were not randomised. Volunteers were allowed to choose the timing of their sampling resulting in some videos, which occurred in different months, being only separated by a few days, while others were more than a month apart. Furthermore, when the timing of diurnal sampling is not consistent (or even randomised) it has the potential to confuse patterns of difference/similarity observed over large time scales (Birt et al., 2012; Harvey et al., 2012a). In regards to spatial biases participants were allowed to choose their sample sites, within each reef, potentially favouring specific sites. This could be problematic due to ‘edge effects’, where a species abundance will vary along the boundary between specific habitats (Ries and Sisk, 2004). With habitat fidelity varying among species it is likely that the over representation of specific clusters will introduce biases, depending on the type of surrounding habitat (Dorenbosch et al., 2005).

A variety of methodology issues relating to the BRUV itself may have also introduced biases into the study. These include potential misidentifications of organisms due to the reduction of light with depth, distance of organisms from the camera and low water quality, making differentiation between morphological characteristics difficult (Kingsford and Battershill, 2000; Widder, 2004). Additionally, the count of individuals may be underestimated when observing high densities of fish, as there is a limit to the number of individuals that can be counted simultaneously (Wraith, 2007). BRUVs have also been noted to underrepresent cryptic species (Watson et al., 2005), site attached fish and those not attracted to bait (Watson et al., 2010). There is also the potential for BRUVs to introduce biases by attracting carnivorous species, influencing the abundance of species that display avoidance behaviour towards large predators (Watson et al., 2010), though this is debated (Willis et al., 2000; Watson et al., 2005; Langlois et al., 2010). As a result, these issues could have potentially introduced bias into the fish community composition recorded, though this would likely be the case regardless of the methodology chosen.

A systemic limitation of the BRUV method is its inability to provide accurate densities, due to difficulties in determining the area sampled by the bait plume (Priede and Merrett, 1996). BRUVs provide relative densities and replicate samples based upon a standardised unit of time, assuming that each deployment is sampling a similar area per unit of time (Taylor et al., 2013). Considering the variable nature of water movements, both between and within the artificial reefs, it can be assumed that bait plume dispersal will change depending on the environmental conditions, resulting in different areas sampled between and within studies (Taylor et al., 2013). To standardise sampling in the method would require the ability to calculate bait plume dispersal, however, given the number of different parameters involved this would be extremely difficult (Harvey et al., 2011). Taylor et al. (2013) showed that when accounting for bait plume dispersal, with rudimentary calculations, community composition results could vary significantly compared to unstandardised results. Therefore, it is likely that each sample within this study is sampling different sized areas, potentially obscuring differences observed.

The methodology used in this study of BRUV based sampling, run through citizen science, has been shown to provide useful monitoring data on the fish communities of artificial reefs in a cost effective manner. Further to that, the results appear to reinforce trends observed over both artificial and natural reefs, indicating that the method has provided relatively accurate results. However, as previously noted, the method is likely to only record a subset of the fish communities present on the artificial reefs. Therefore, it is important that if this methodology is to be utilised in future studies that this limitation is acknowledged or addressed by supplementing the program with additional monitoring through other methods. This approach has been adopted by the Department of Fisheries (Western Australia) who have conducted BRUV based sampling whilst also utilising Diver Operated Video (DOV) when monitoring the fauna of the Geographe Bay artificial reefs (Appendices 1).

Despite the methodologies limitations, which are inherent in any monitoring method, the Reef Vision program has generated a large and useful data set that would prove to be cost-prohibitive to collect through other means. Therefore, utilising BRUV based sampling, deployed through citizen science, should be considered an option for monitoring the fish communities of artificial reefs in the future.

3.5.4.2. Future work

Due to time constraints, only nine months of data could be analysed in the current study. Thus, the first step in furthering the work presented here would be to analyse data collected for August and September. This analysis would allow for the determination of whether the patterns observed during the winter months are typical or driven by short-term events. This is particularly important as, due to poor weather and thus a lack of videos, June and July had to be merged into a single sample, potentially obscuring subtle trends between these months and winter in general.

While the methodology employed here resulted in the collection of sufficient data, developing a more regimented approach to the citizen science monitoring program would help to remove some temporal and spatial biases. However, it was noted by Florisson (2015) that increased requirements placed upon participants, *i.e.* sampling a natural reef in addition to the artificial reefs, resulted in low volunteer participation and thus limited quantities of data being collected. This is supported by the literature with highly regimented citizen science programs being found to result in low recruitment and retention rates of volunteers (Dickinson et al., 2010). One modification to reduce spatial bias in data collection, without increasing regulation significantly, could be to have participants deploy their BRUVs on specific clusters. Although controlling the timing of sampling, *i.e.* asking volunteers to all sample on specific days, may make the program too regimented and lower participation rates.

While this study has investigated the characteristics of the fish communities of the artificial reefs in Geographe Bay, it provides no comparison with adjacent natural reefs or sand/seagrass areas. This could be used to assess how effectively the reefs mimic nearby natural reefs, provide further investigation into the effect of habitat connectivity, as well as helping to determine whether the fish communities of these artificial reefs are influenced by ‘attraction’ and/or ‘production’. Ultimately, these investigations would allow for an evaluation of the performance of the reef against its stated goals and contribute to the broader scientific literature on artificial reefs. To achieve this, utilising the current methodology, would require significant alteration of the current program and again may hinder participation rates. Alternatively, a concurrent BRUV based citizen science program, or another program utilising different methodology, could be created to solely sample natural reefs.

Standardising sampling among videos by calculating the area of the bait plume would also provide a significant improvement to the current methodology. The addition of a current meter to the BRUV frame would allow for the measurement of the strength and direction of currents on each deployment to be recorded, providing a rough estimation of the area covered by the plume and thus enabling the standardisation of each video drop (Harvey et al., 2011).

Furthermore, with the limitations inherent to BRUVs (see Chapter 2), it is likely that the monitoring program utilised only recorded a subset of the fish communities present on the reefs. A more accurate representation of the fish fauna could be recorded with the addition of another technique, such as an underwater visual census method. Given that the current monitoring program was developed to be cost-effective, the second method could be employed less regularly (*e.g.* seasonally or annually) and/or deployed/conducted using citizen science.

Chapter 4: General conclusion

Over the past five years the number of artificial reefs in Australia has expanded greatly, with the majority being deployed to increase the localised abundance of fish (Bateman, 2015). This increase is believed to be due to the provision of additional habitat, which results in their increased production and/or attraction and concentration from the wider environment (Bohnsack and Sutherland, 1985; Baine, 2001; Simon et al., 2011). It is this alteration of the surrounding environment that has led to legislation potentially requiring the biological monitoring of artificial reef deployments to ensure any negative impacts are identified and ameliorated (Department of Fisheries, 2012; Department of the Environment, 2016b).

With a number of artificial reef deployments planned around Western Australia, as well as those already underway, it is essential that cost-effective faunal monitoring programs be developed (Recfishwest, 2016). As such, Recfishwest, through funding from the FRDC, has instigated a number of research projects on artificial reefs, including the development of cost-effective methods for their monitoring. As part of this research, this thesis has reviewed a range of techniques for monitoring the fauna of artificial reefs and applied one of these, the BRUV system, in a citizen science program (Reef Vision) to monitor the characteristics of the Bunbury and Dunsborough artificial reefs.

Firstly, a critical review of a wide variety of methods for monitoring the faunas of artificial reefs, from traditional underwater visual census techniques to rapidly emerging fields such as environmental DNA analysis, was conducted (Chapter 2). Each method was assessed against five criteria; deployment, accuracy, precision, time and cost. The review found that the effectiveness of each monitoring method was dependant on a number of factors including, the type of fauna targeted, the type of artificial reef being monitored, the spatial scale of monitoring required and other considerations, such as logistics, budget and the timeframe of the monitoring program. Therefore, the type of monitoring method chosen should be based upon the objectives and logistical considerations of each study. One technique that scored well was the remotely operated underwater video method, which includes BRUV. This method was found to be relatively inexpensive and easy to deploy, whilst providing a moderate level of species richness and abundance accuracy of mobile fauna. Furthermore, the method was found to be effective for use with citizen science (Bear, 2016; Raoult et al., 2016).

Building on the earlier work by Florisson (2015) and Bateman (2015), this study investigated whether the characteristics of the fish faunas of the Bunbury and Dunsborough artificial reefs differed both spatially and temporally by utilising BRUV (Chapter 3). Local recreational fishers were recruited to deploy the BRUV units on the artificial reefs providing samples from October 2015 to July 2016. The results of this project found that the reefs showed significant differences geographically, as well as temporally, with a number of variables. The factor reef was found to have the most influence within these variables, indicating that habitat connectivity played a large part in determining the associated fish communities of the reefs. Significant results were also found with the factors month and the interaction between the main effects, suggesting that changes in temperature, as well as oceanographic processes, also contribute to shaping the reefs fish communities. Therefore, from these results it can be understood that the fish communities of the artificial reefs of Geographe Bay differ geographically and show temporal variations.

Furthermore, this study has demonstrated that the BRUV technique, deployed through citizen science, can be used successfully for monitoring the fish faunas of artificial reefs. With future deployments of artificial reefs, both planned and already underway around Western Australia, this program would be able to provide cost-effective monitoring for their faunas, particularly in regional areas where Government and tertiary institution based work can be financially prohibitive (Recfishwest, 2016).

To summarise, this thesis has provided an evaluation of the various fauna monitoring techniques available for artificial reefs, and then used one of these techniques, BRUV, to study the fish faunas of the artificial reefs of Geographe Bay. Ultimately, it has shown these fish communities to differ both geographically, and temporally, highlighting the usefulness of the technique in monitoring artificial reefs.

Appendices

Appendices 1. Fish species recorded over different studies of the Bunbury and Dunsborough artificial reefs in Geographe Bay. Studies include: this thesis which utilised Baited Remote Underwater Video (BRUV), deployed through citizen science, between October 2015 and June/July 2016; BRUV and Diver Operated Video monitoring by the Department of Fisheries (Western Australia) over six surveys between April 2013 to October 2014; and studies by Bateman (2015), utilising BRUV across three sampling dates in March and May 2015, and Florisson (2015), using BRUV between November 2014 and April 2015. A tick indicates that the species was found in these studies.

Taxa	Thesis	Department of Fisheries	Bateman (2015) and Florisson (2015)
Achoerodus gouldii	✔	✔	✔
Anoplocapros amygdaloides	✔	✔	✔
Anoplocapros lenticularis	✔	✔	✔
Apogon victoriae	✔	✔	
Aptychotrema vincentiana	✔	✔	✔
Aracana aurita	✔	✔	✔
Argyrosomus japonicus	✔		
Arripis truttaceus	✔		
Aulohalaelurus labiosus		✔	
Austrolabrus maculatus	✔	✔	✔
Chaetodon assarius	✔		
Caesioscorpis theagenes		✔	
Cheilodactylus gibbosus	✔	✔	✔
Cheiodactylus nigripes			✔
Chelmonops curiosus	✔	✔	✔
Choerodon rubescens	✔	✔	✔
Chromis klunzingeri		✔	✔
Chromis westaustralis	✔		
Coris auricularis	✔	✔	✔
Dactylophora nigricans	✔	✔	
Dasyatis brevicaudata	✔	✔	✔
Diodon nicthemerus	✔	✔	✔
Enoplosus armatus	✔		
Eubalichthys mosaicus	✔	✔	✔
Glaucosoma hebraicum	✔	✔	✔
Haliocheres brownfieldi	✔	✔	
Hypoplectrodes nigroruber	✔		
Haliochoeres brownfieldii		✔	
Helcogramma decurrens		✔	
Heniochus acuminatus		✔	
Hypoplectrodes nigroruber		✔	
Meuschenia freycineti		✔	✔
Meuschenia venusta	✔		
Monocanthidae spp.	✔		
Mustelus antarcticus	✔	✔	
Myliobatus antarcticus		✔	
Myliobatus australis	✔	✔	✔
Neatypus obliquus	✔	✔	✔
Neosebastes pandus		✔	✔
Eupetrichthys angustipes	✔	✔	
Notolabrus parilus	✔	✔	
Octopus tetricus	✔		
Ophthalmolepis lineolatus	✔	✔	
Orectolobus maculatus	✔		
Chrysophorus auratus	✔	✔	✔
Parapercis haackei	✔	✔	✔
Parapercis ramsayi	✔		
Paraplotosus albilabris		✔	
Parapriacanthus elongatus	✔	✔	
Parequula melbournensis	✔	✔	✔
Paristiopterus gallipavo		✔	
Parma mccullochi		✔	
Parupeneus chrysopleuron	✔	✔	
Pempherididae spp.	✔	✔	✔
Pentapodus vittae		✔	
Pempheris klunzingeri	✔	✔	✔
Pentaceropsis recurvirostris	✔		✔
Platycephalus longispinis	✔	✔	

Taxa	Thesis	Department of Fisheries	Bateman (2015) and Florisson (2015)
Platycephalus sp.		✔	
Platycephalus speculator	✔	✔	
Pseudocaranx spp	✔	✔	
Pseudolabrus biserialis		✔	✔
Pseudorhombus jenynsii	✔	✔	
Rhabdosargus sarba	✔		
Scobinichthys granulatus	✔		
Sepia apama	✔		
Sepioteuthis australis	✔		
Seriola hippos	✔	✔	✔
Siganus fuscescens	✔		✔
Siganus sp.		✔	
Suezichthys cyanolaemus	✔		
Tilodon sexfasciatus	✔	✔	✔
Trachinops noarlungae		✔	✔
Trachurus novaezelandiae	✔	✔	✔
Trygonoptera mucosa	✔		
Trygonoptera ovalis	✔		
Trygonoptera personata	✔	✔	✔
Trygonorrhina dumerilli	✔	✔	✔
Upeneichthys vlamingii	✔	✔	✔
Urolophus sp.		✔	✔
Total no. of taxa detected	**60**	**59**	**36**

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